Greg A. Hibbard, OSB No. 183602
ghibbard@nwresourcelaw.com
NORTHWEST RESOURCE LAW PLLC
71 Columbia Street, Suite 325
Seattle, WA 98104
206.971.1568

David O. Bechtold, OSB No. 133019
dbechtold@ nwresourcelaw.com
NORTHWEST RESOURCE LAW PLLC
1500 SW First Avenue, Suite 985
Portland, OR 97201
503.664.3582

Attorneys for Proposed Amicus Curiae
James Johnston, Ph.D.

UNITED STATES DISTRICT COURT

DISTRICT OF OREGON

PENDLETON DIVISION

GREATER HELLS CANYON COUNCIL, et al.,

Plaintiffs,

v.

HOMER WILKES, et al.,

Defendants,

and

AMERICAN FOREST RESOURCE COUNCIL and EASTERN OREGON COUNTIES ASSOCIATION,

Defendant-Intervenors.

CV No. 2:22-CV-00859-HL

DECLARATION OF JAMES JOHNSTON, Ph.D.

James Johnston, Ph.D., declares as follows:

1. I am over 18 years of age and submit this declaration in support of the *amicus curiae* brief filed on my behalf. I have personal knowledge of the matters stated herein and, if called as a witness, could and would competently testify thereto.

2. I am an Assistant Professor in the College of Forestry at Oregon State University. I have received two post-graduate degrees from Oregon State University: a Master of Science in Forest Resources and a Doctor of Philosophy in Forest Science. I have research experience with historical forest conditions, natural and anthropogenic disturbances of forest health, and interior dry and mixed conifer forests. As with the rest of my involvement with this proposed *amicus curiae* brief, I make this declaration in my personal capacity.

3. Attached as **Exhibit A** to my declaration is a true and complete copy of a letter I have written to the Court, with feedback from some of my colleagues, that expresses my opinions on Plaintiffs' arguments pertaining to scientific controversy in this matter. Collectively, the signatories have been at the forefront of forest science in the Pacific Northwest for decades. Each of the additional signors are widely respected forest scientists, many of whom work at renowned universities or well-known public interest organizations. Although listed with their current professional titles, each has signed only in their individual capacity and does not speak on behalf of their employer with this letter. In addition to myself, those signing in agreement with the letter include:

1. Derek Churchill, Ph.D., Forest Health Scientist, Washington State Department of Natural Resources
2. Don Falk, Ph.D., Professor, University of Arizona School of Natural Resources and

NORTHWEST RESOURCE LAW PLLC
1500 SW First Avenue, Suite 985
Portland, OR 97201
503.664.3582

the Environment

3. Jerry Franklin, Ph.D., Professor Emeritus, College of Forest Resources at University of Washington
4. Keala Hagmann, Ph.D., Research Ecologist, Applegate Forestry LLC
5. Lori D. Daniels, Ph.D., Professor, Department of Forest and Conservation Sciences at the University of British Columbia
6. Matthew Hurteau, Ph.D., Professor, Department of Biology at the University of New Mexico
7. Meg Krawchuk, Ph.D., Associate Professor, College of Forestry at Oregon State University
8. Norm Johnson, Ph.D., Professor Emeritus, College of Forestry at Oregon State University
9. Peter M. Brown, Ph.D., Director, Rocky Mountain Tree-Ring Research
10. Robert W. Gray, Fire Ecologist, R.W. Gray Consulting, Ltd.
11. Scott Stephens, Ph.D., Professor of Fire Science, University of California Berkeley
12. Susan Prichard, Ph.D., Fire Ecologist, University of Washington School of Environmental and Forest Sciences
13. Thomas H. Deluca, Ph.D., Dean, College of Forestry at Oregon State University
14. Trent Seager, Ph.D., Director of Science, Sustainable Northwest

NORTHWEST RESOURCE LAW PLLC
1500 SW First Avenue, Suite 985
Portland, OR 97201
503.664.3582

Pursuant to 28 U.S.C. § 1746, I declare under penalty of perjury that the foregoing is true and correct.

Executed on: February 6, 2023.

James Johnston, Ph.D.

NORTHWEST RESOURCE LAW PLLC
1500 SW First Avenue, Suite 985
Portland, OR 97201
503.664.3582

I hereby certify that I electronically filed the foregoing with the Clerk of the Court for the United States District Court, District of Oregon, Pendleton Division using the CM/ECF system on February 10, 2023. I further certify that all participants in the case are registered CM/ECF users and that service will be accomplished by the CM/ECF system.

DATED this 10th day of February, 2023.

s/ Greg A. Hibbard
Greg A. Hibbard, OSB No. 183602
David O. Bechtold, OSB No. 133019

Attorneys for Proposed Amicus Curiae James Johnston, Ph.D.

NORTHWEST RESOURCE LAW PLLC
1500 SW First Avenue, Suite 985
Portland, OR 97201
503.664.3582

Exhibit A

February 1, 2023

Honorable Andrew D. Hallman
United States District Court, District of Oregon, Pendleton Division
104 S.W. Dorion Avenue
Pendleton, OR 97801

Re: ***Greater Hells Canyon Council, et al. v. Homer Wilkes, et al.***
USDC District of Oregon Case No. 2:22-CV-00859-HL

Dear Honorable Andrew D. Hallman:

We are forest ecologists who have conducted extensive empirical and theoretical research that describes historical forest conditions, changes to forests over time, and the effects of fire, fire exclusion, logging, and other disturbances on forests. Although we write to the Court in our personal capacity, we work for public universities and non-governmental organizations (federal agency policies prohibited our colleagues employed by the Forest Service from participating in this brief). Much of our research has been conducted in interior dry and moist mixed conifer forests. Many of us work closely with Forest Service managers and community stakeholders to design and implement restoration projects and monitor the ecological outcomes of these projects. Some of us consulted extensively with the Forest Service in development of the amendment in question and several of us prepared detailed comments on the draft Environmental Assessment (EA) for the Screens Amendment.

We are submitting this letter to address concerns raised by Plaintiffs about scientific controversy. As we explain below, although a handful of independent researchers make a number of claims that give the appearance of controversy, there is no meaningful controversy among the scientific community with respect to changes to forests over time or the effects of common restoration actions.

Some of us recently completed three invited review articles as part of a special issue of *Ecological Applications* that speak directly to the issue of scientific controversy raised by Plaintiffs. We have included those papers with this letter for the Court's consideration.

I. The eastside screens and the 21-inch rule amendment.

In 1995, the Forest Service adopted an amendment to forest plans governing the management of national forests in eastern Oregon and Washington that were outside the Northwest Forest Plan area. This amendment, commonly referred to as the "21-inch rule," stated:

> *Outside of LOS[*], many types of timber sale activities are allowed. The intent is still to maintain and/or enhance LOS components in stands subject to timber harvest as much as possible, by adhering to the following standards: a) Maintain all remnant late and old seral and/or structural live trees ≥21-inch dbh that currently exist within stands proposed for harvest activities.*

The amendment was designed to be in place for 12 to 18 months until adoption of a new ecosystem management plan. That plan was written but never formally adopted, and the 21-inch rule remained in place until 2021. In the 25+ years since the 21-inch rule was adopted, much has changed. The number of trees >21 inches diameter at breast height (DBH) has increased by 17% (EA at 102). Mortality of old trees and degradation of valuable habitat has increased significantly as a direct consequence of increasing density of forests and drought, wildfire, and insect disturbance (EA at 104-105). In response, in January 2021 the Forest Service adopted a new amendment governing management of 7.9 million acres on the Fremont-Winema, Deschutes, Ochoco, Malheur, Umatilla, and Wallowa-Whitman national forests ("eastside forests"). This amendment replaces the 21-inch rule and states:

> *Outside of LOS, many types of timber sale activities are allowed. The intent is still to maintain and/or enhance a diverse array of LOS conditions in stands subject to*

[*] LOS stands for late and old structure and refers to forests with older trees.

> *timber harvest as much as possible, by adhering to the following plan components: Managers should retain and generally emphasize recruitment of old trees and large trees, including clumps of old trees. Management activities should first prioritize old trees for retention and recruitment. If there are not enough old trees to develop LOS conditions, large trees should be retained, favoring fire tolerant species where appropriate. Old trees are defined as having external morphological characteristics that suggest an age ≥150 years. Large trees are defined as grand fir or white fir ≥30 inches dbh or trees of any other species ≥21 inches dbh. Old and large trees will be identified through best available science. Management activities should consider appropriate species composition for biophysical environment, topographical position, stand density, historical diameter distributions, and spatial arrangements within stands and across the landscape in order to develop stands that are resistant and resilient to disturbance.*

In essence, the blanket prohibition on cutting trees >21 inches DBH has been replaced by a guideline that emphasizes protection of old trees (defined as trees ≥150 years of age) and protection of large trees (defined as grand fir ≥30 inches DBH and all other species ≥21 inches DBH).

II. Alleged scientific controversy about historical forest conditions and effects of the new amendment.

Plaintiffs allege that replacement of the 21-inch rule with a new rule for protection of large and old trees is "highly controversial and will yield uncertain effects" (Plaintiffs' Motion for Summary Judgement (MSJ) at iii). Many of Plaintiffs' arguments about scientific controversy (Plaintiff's Memorandum in Support of MSJ at 25-36) appear to be a straw man designed to confuse salient issues rather than accurately characterize the state of the science. We believe that Plaintiffs' arguments are designed to give the impression of scientific controversy where no meaningful controversy among scientists exists. As we wrote in a recent invited review in *Ecological Applications* ("Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests", Hagmann et al. 2021):

> Perpetuating invalidated methods and inferences based on them fosters confusion and controversy, which undermine scientific credibility and impede the development of relevant and timely policy and management options. … Objective scientific evaluation can aid in differentiating warranted from unwarranted uncertainties and enable timely paradigm shifts to policies and management actions that favor fire- and climate-adapted forests and human communities.

Below, we briefly discuss several issues that we believe are important from the standpoint of the scientific literature about changes to forests over time and the effects of forest restoration treatments.

A. The historical condition of eastside forests.

The overwhelming majority of research and observations show conclusively that eastside Oregon and Washington forests have undergone dramatic ecological changes and that these changes have left valuable resources vulnerable to uncharacteristic disturbances including: old-growth trees; wildlife habitat; vital water supplies and culturally important food; medicinal resources; and material resources for Indigenous tribes (as discussed by the Klamath Tribe in its comments on the Screens Amendment EA). In our view, the Forest Service in the EA and decision notice (DN) made reasonable conclusions from the scientific literature about the historical condition of eastside forests, changes to these forests over time, and the need for restoration.

In their Memorandum in Support of MSJ, Plaintiffs refer to a non-peer reviewed report from Drs. DellaSala and Baker (the DellaSala/Baker report). Plaintiffs claim this report shows that "some researchers and the agency [] falsely conclude that Eastside forests were predominately open park-like pine forests, when, in fact, fire regimes and forest structure and composition were much more complex;" and "[d]enser closed canopy forests were more prevalent, and shade tolerant trees were more common… than

acknowledged in the EA" (Memorandum in Support of MSJ at 28 (quoting DellaSala/Baker report) (alterations in original)).

These claims are specious. The plain language of the Forest Service's amendment does not contemplate returning all or most extant eastside forests to an open, park-like condition. To the best of our knowledge, the EA does not claim that eastside forests historically consisted entirely, or for the most part, of open park-like pine forests. The research that we authored and that was cited in the EA does not claim that eastside forests consisted entirely, or for the most part, of open park-like forests. To the best of our knowledge, the EA does not claim, and our research and our colleagues' research cited in the EA does not claim, that shade-tolerant trees were not a component of historical stands.

On the contrary, our peer-reviewed work and the work cited in the EA acknowledges significant variability in historical forest structure and composition, including the presence of shade-tolerant grand fir in some historical stands. The Forest Service's EA correctly points out that although there was significant variability in historical forests, the number of shade-tolerant trees and the proportion of shade-tolerant trees as a percentage of total stand biomass have increased significantly in the past 150 years (see our research and colleagues' research cited in the EA, including Hagmann et al. 2021, Hessburg et al. 2021, Johnston et al. 2021, Hessburg et al. 2020, Lindsay and Johnston 2020, Hagmann et al. 2019, Heyerdahl et al. 2019, Merschel et al. 2018, Hagmann et al. 2018, Johnston et al. 2018, Johnston 2017, Hagmann et al. 2014, Merschel et al. 2014, Hagmann et al. 2013).

Plaintiffs claim that the non-peer reviewed report by Drs. DellaSala and Baker "explains in detail the problems with major studies the agency relies on to support its

decision and highlights the scientific debate" (Memorandum in Support of MSJ at 29). The DellaSala/Baker report is full of mischaracterizations of other scientists' research and contains no meaningful theoretical or empirical rebuttal of our findings or our colleagues' findings. We believe the major point of the DellaSala/Baker report is simply to confuse the reader. Mischaracterizing other researchers' work and then attacking that mischaracterization is in the nature of knocking down a straw man and does not demonstrate the existence of meaningful scientific controversy. The tactics of this report—a barrage of critiques not rooted in the substance of the scientific literature—have been analyzed in two recent papers designed to help policymakers distinguish scientific misinformation from accurate and actionable science. We recommend these papers (Jones et al. 2022 and Peery et al. 2019, see citations below) to the Court.

Plaintiffs allege that "other data sources (Baker (2012), Hessburg et al (2007), others) made contrary findings regarding the historical presence of grand firs" (Memorandum in Support of MSJ at 29). This argument is a straw man. The scientific literature cited in the EA does not deny that grand fir was historically present on the landscape; it simply states that there is significantly more grand fir today than existed historically. Hessburg et al. 2007, according to the author, does not contradict the findings of the EA (Hessburg et al. 2020, Spies et al. 2018, Stine et al. 2014). The EA explicitly acknowledges that the methods presented by Baker (2012) produce different findings about the extent of grand fir in historical stands than the vast majority of peer-reviewed studies. The EA also correctly points out that a variety of studies have shown Baker's methodology to overestimate historical forest density (EA at 57; *see also* e.g., Hagmann et al. 2021, Levine et al. 2019, Johnston et al. 2018, Levine et al. 2017).

In summary, although it is clear that Drs. Baker, DellaSala, and others do not accept the scientific consensus about historical eastside forest conditions, this does not imply that there is meaningful or significant scientific controversy. As we wrote in a recent synthesis ("Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests", Hagmann et al. 2021) that summarizes the state of the science about historical forests and changes to forests over time:

> Based on the strength of evidence, there can be little doubt that the long-term deficit of abundant low- to moderate-severity fire has contributed to modification of seasonally dry forested landscapes across western North America. The magnitude of change in fire regimes and the resultant increases in forest density and fuel connectivity have increased the vulnerability of many contemporary forests to seasonal and episodic increases in drought and fire, exacerbated by rapid climate warming.

B. Effects of the new amendment.

Plaintiffs assert that the effects of implementing the new amendment are highly uncertain because the modeling effort undertaken by the Forest Service is "fatally flawed" (Memorandum in Support of MSJ at 33). They believe the Forest Service's modeling effort is fatally flawed in large part because we ourselves (Drs. Franklin, Johnson, and Seager), in comments on the Forest Service's draft EA, asked the Forest Service to clarify typical tree retention during thinning operations. The Forest Service offered a clarification in the final EA, and we believe that the modeling exercise described in the final EA provides a reasonable basis for evaluating environmental effects of silviculture that makes use of the new amendment. The Forest Vegetation Simulator (FVS) has been the tool of choice for several decades when estimating effects of thinning and other harvests on resulting stand conditions and their vulnerability to disturbances.

Throughout their motion for summary judgment, Plaintiffs imply that replacing a blanket prohibition on cutting trees >21 inches DBH with a guideline that emphasizes protection of old trees (all trees ≥150 years of age) and large trees (grand fir ≥30 inches DBH and all other species ≥21 inches DBH) simultaneously involves considerable uncertainty and significant environmental impacts. We are aware of little or no meaningful debate or controversy in the scientific literature about the importance of large and old trees. Crafting guidelines for the protection of large and old trees was among our explicit recommendations in the attached review in *Ecological Applications* (Hessburg et al. 2021) (internal citations omitted):

> Most research reveals that broadly conserving large and old fire-resistant trees and replacing those that were removed or killed by harvest, drought, insects, pathogens, and wildfires provides a strong backbone of resilient structure and habitat to seasonally dry pine and mixed-conifer ecosystems.

We have participated in the design and implementation of dozens of Forest Service projects that have thinned tens of thousands of acres of forest, including treatments that removed grand fir that are ≥21 inches DBH but younger than 150 years of age. In general, the effects of these treatments are relatively predictable and reduce uncertainty with respect to future disturbances and effects of disturbances on wildlife habitat, water quality, and carbon stocks. We do not believe that the Forest Service's EA or decision notice mischaracterized the state of the science relevant to environmental effects of these treatments or made unreasonable assumptions.

In their objections to the new guidelines, Plaintiffs are essentially requesting that the court maintain the *status quo ante* of forest management and forest conditions. We explicitly address claims made by Plaintiffs about uncertainty in effects of restoration and the relative risks of restoration vs. no-action in two recent invited syntheses ("Adapting

western North American forests to climate change and wildfires: 10 common questions" and "Wildfire and climate change adaptation of western North American forests: a case for intentional management"; Hessburg et al. 2021, Prichard et al. 2021). We wrote:

> The precautionary principle can become the "paralyzing principle" given irreducible uncertainty about risk of loss associated with action and no-action alternatives (Sunstein 2003). The loss of 30 million mature and old pine trees during a recent extreme drought in south-central California (Asner et al. 2016) is a stark reminder of the pitfall of requiring unduly high certainty despite decades of established science showing the efficacy of treatments that foster resilient forest structure and composition (Henson et al. 2018, Fettig et al. 2019).
>
> …
>
> Scientific knowledge is always growing and incomplete. However, a preponderance of evidence suggests that proactive management can prepare many landscapes for future wildfires and the maintenance work they can provide. This would also reduce emphasis on high-maintenance solutions and the overarching and increasingly burdensome role of wildfire suppression and its expenditures.

Literature Cited

Baker, W.L. 2012. Implications of Spatially Extensive Historical Data from Surveys for Restoring Dry Forests of Oregon's Eastern Cascades." Ecosphere 3(3) https://doi.org/10.1890/ES11-00320.1.

Hagmann R.K., Hessburg P.F., Prichard S.J., Povak N.A., Brown P.M., Fulé P.Z., Keane R.E., Knapp E.E., Lydersen J.M., Metlen K.L., Reilly M.J., Sánchez Meador A.J., Stephens S.L., Stevens J.T., Taylor A.H., Yocom L.L., Battaglia M.A., Churchill D.J., Daniels L.D., Falk D.A., Krawchuk M.A., Johnston J.D., Levine C.R., Meigs G.W., Merschel A.G., North M.P., Safford H.D., Swetnam T.W., Waltz A.E.M. 2021. Evidence for Widespread Changes in the Structure, Composition, and Fire Regimes of Western North American Forests. Ecological Applications 31(8) https://doi.org/10.1002/eap.2431

Hagmann, R.K., Merschel, A.G., and Reilly, M.J. 2019. Historical patterns of fire severity and forest structure and composition in a landscape structured by frequent large fires: Pumice Plateau ecoregion, Oregon, USA. Landscape Ecology 34:551–568.

Hagmann, R.K., Stevens, J.T., Lydersen, J.M., Collins, B.M., Battles, J.J., Hessburg, P.F., Levine, C.R., Merschel, A.G., Stephens, S.L., Taylor, A.H. and Franklin, J.F. 2018. Improving the use of early timber inventories in reconstructing historical dry forests and fire in the western United States: Comment. Ecosphere 9(7).

Hagmann, R.K., Franklin, J.F. and Johnson, K.N. 2014. Historical conditions in mixed-conifer forests on the eastern slopes of the northern Oregon Cascade Range, USA. Forest Ecology and Management, 330: 158-170.

Hagmann, R.K., Franklin, J.F. and Johnson, K.N. 2013. Historical structure and composition of ponderosa pine and mixed-conifer forests in south-central Oregon. Forest Ecology and Management, 304: 492-504.

Hessburg P.F., Prichard S.J., Hagmann R.K., Povak N.A., and Lake F.K. 2021. Invited review: Wildfire and Climate Change Adaptation of Western North American Forests: A Case for Intentional Management. Ecological Applications 31(8) https://doi.org/10.1002/eap.2432

Hessburg P.F., Charnley S., Wendel K.L., White E.M., Singleton P.H., Peterson D.W., Halofsky J.E., Gray A.N., Spies T.A., Flitcroft R.L., White R. 2020. The 1994 Eastside Screens – Large Tree Harvest Limit: Review of Science Relevant to Forest Planning 25 years Later. Gen. Tech. Rep. PNW-GTR-990. Portland, OR: USDA-FS, Pacific Northwest Research Station: 124 p.

Hessburg, P.F., Salter, R.B., and James, K.M. 2007. Reexamining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. Landscape Ecology 22:5–24.

Heyerdahl, E.K., Loehman, R.A. and Falk, D.A. 2019. A multi-century history of fire regimes along a transect of mixed-conifer forests in central Oregon, USA. Canadian Journal of Forest Research 49(1): 76-86.

Johnston, J.D., Greenler, S.M., Miller, B.A., Reilly, M.J., Lindsay, A.A., and Dunn, C. J. 2021. Diameter limits impede restoration of historical conditions in dry mixed-conifer forests of eastern Oregon, USA. Ecosphere 12(3), e03394.

Johnston, J.D., Dunn, C.J., Vernon, M.J., Bailey, J.D., Morrissette, B.A. and Morici, K.E. 2018. Restoring historical forest conditions in a diverse inland Pacific Northwest landscape. Ecosphere 9(8): e02400.

Johnston, J.D. 2017. Forest succession along a productivity gradient following fire exclusion. Forest Ecology and Management 392: 45-57.

Jones, G.M., Vraga, E. K., Hessburg, P.F., Hurteau, M.D., Allen, C.D., Keane, R.E., Spies, T.A., North, M.P., Collins, B.M., Finney, M.A., Lydersen, J.M., and Westerling, A.L. 2022. Counteracting wildfire misinformation. Frontiers in Ecology and the Environment 20:392-393 https://doi.org/10.1002/fee.2553

Levine, C.R., Cogbill, C.V., Collins, B.M., Larson, A.J., Lutz, J.A., North, M.P., ... & Battles, J.J. 2017. Evaluating a new method for reconstructing forest conditions from General Land Office survey records. Ecological Applications, 27(5), 1498-1513.

Levine, C.R., Cogbill, C.V., Collins, B.M., Larson, A.J., Lutz, J.A., North, M.P., Restaino, C.M., Safford, H.D., Stephens, S.L., and Battles, J.J. 2019. Estimating historical forest density from land-survey data: A response to Baker and Williams (2018). Ecological Applications 29:e01968.

Lindsay, A.A., & Johnston, J.D. 2020. Using historical reconstructions of moist mixed conifer forests to inform forest management on the Malheur National Forest. In In: Pile, Lauren S.; Deal, Robert L.; Dey, Daniel C.; Gwaze, David; Kabrick, John M.; Palik, Brian; Schuler, Thomas M., comps. The 2019 National Silviculture Workshop: a focus on forest management-research partnerships. Gen. Tech. Rep. NRS-P-193. Madison, WI: US Department of Agriculture, Forest Service, Northern Research Station: 23-33. (pp. 23-33).

Merschel, A.G., Heyerdahl, E.K., Spies, T.A., and Loehman, R.A. 2018. Influence of landscape structure, topography, and forest type on spatial variation in historical fire regimes, Central Oregon, USA. Landscape Ecology 33(7):1195-1209

Merschel, A.G., Spies, T.A., & Heyerdahl, E.K. 2014. Mixed-conifer forests of central Oregon: effects of logging and fire exclusion vary with environment. Ecological Applications 24(7):1670-1688

Peery, M.Z., Jones, G.M., Gutiérrez, R., Redpath, S.M., Franklin, A.B., Simberloff, D., Turner, M.G., Radeloff, V.C., and White, G.C. 2019. The conundrum of agenda-driven science in conservation. Frontiers in Ecology and Evolution. https://doi.org/10.1002/fee.2006

Prichard, S.J., Hessburg P.F., Hagmann R.K., Churchill D.J., Povak N.A., Dobrowski S., Gray R.W., Huffman D., Hurteau M.D., Kane V., Pratima K.C., Keane R.E., Kobziar L., Lake F.K., North M.P., Safford H., Stevens J., Kolden C., Parks S., and Yocom L. 2021. Invited review: Adapting Western North American Forests to Climate Change and Wildfires: Ten common questions. Ecological Applications https://doi.org/10.1002/eap.2433

Spies, T.A., Hessburg, P.F., Skinner, C.N., Puettmann, K.J., Reilly, M.J., Davis, R. J., ... & Shaw, D. C. (2018). Old growth, disturbance, forest succession, and management in the area of the Northwest Forest Plan. *Synthesis of science to inform land management within the Northwest Forest Plan area*, *1*

Stine, P., Hessburg, P.F., Spies, T.A., Kramer, M.G., Fettig, C.J., Hansen, A.J., ... & White, R. 2014. Ecology and Management of Moist Mixed-conifer Forests in Eastern Oregon and Washington: A Synthesis of the Relevant Biophysical Science and Implications for Future Land Management. United States Department of Agriculture, Forest Service, Pacific Northwest Research Station

We, the undersigned, agree with and submit the contents of this letter for the Court's consideration. Although listed with our current professional titles, we have signed only in our individual capacities and do not speak on behalf of our employers with this letter. Signatures are included on the following pages.

Derek Churchill, Ph.D., Forest Health Scientist, Washington State Department of Natural Resources

Don Falk, Ph.D., Professor, University of Arizona School of Natural Resources and the Environment

James Johnston, Ph.D., Assistant Professor, College of Forestry at Oregon State University

Jerry Franklin, Ph.D., Professor Emeritus, College of Forest Resources at University of Washington

Keala Hagmann, Ph.D., Research Ecologist, Applegate Forestry LLC

Lori D. Daniels, Ph.D., Professor, Department of Forest and Conservation Sciences at the University of British Columbia

Matthew Hurteau, Ph.D., Professor, Department of Biology at the University of New Mexico

Meg Krawchuk, Ph.D., Associate Professor, College of Forestry at Oregon State University

Norm Johnson, Ph.D., Professor Emeritus, College of Forestry at Oregon State University

Peter M. Brown, Ph.D., Director, Rocky Mountain Tree-Ring Research

Robert W. Gray, Fire Ecologist, R.W. Gray Consulting, Ltd.

Scott Stephens, Ph.D., Professor of Fire Science, University of California Berkeley

Susan Prichard, Ph.D., Fire Ecologist, University of Washington School of Environmental and Forest Sciences

Thomas H. Deluca, Ph.D., Dean, College of Forestry at Oregon State University

Trent Seager, Ph.D., Director of Science, Sustainable Northwest

James Johnston

James Johnston, Ph.D.

Date: 2/1/23

Trent Seager, Ph.D.

Date: 5 Feb 2023

Susan Prichard, Ph.D.

Date: Feb 5, 2023

Jerry J. Franklin

Jerry Franklin, Ph.D.

Date: February 3, 2023

Norm Johnson

Norm Johnson, Ph.D.

Date: 2/4/2023

Keala Hagmann, Ph.D.

Date: 2/6/2023

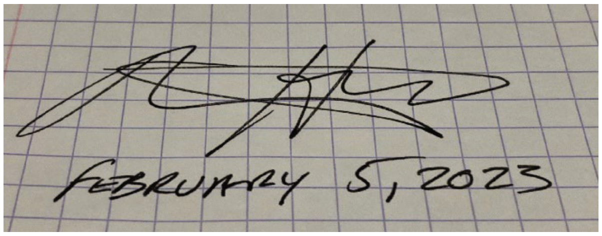
Robert Gray
____

Robert W. Gray

Date: February 7, 2023

Lori Daniels, Ph.D.

Date: February 6, 2023



Matthew Hurteau, Ph.D.

As requested by Matthew Hurteau, Ph.D., I affixed the signature above.

/s Eliza Hinkes
Eliza Hinkes, Paralegal

Name: Scott Stephens

Date: 2/7/2023

Thomas H. DeLuca

Thomas H. DeLuca, Ph.D.

Date: 02/02/2023

____

Name: Peter M. Brown

Date: Feb 3 2023

[signature]

Name: Derek Churchill

Date: 2/6/2023

Name: Donald A Falk

Date: 6 February 2023

M. Krawchuk

Name: Meg A. Krawchuk, Ph.D.

Date: 02/03/2023

Ecological Applications, 31(8), 2021, e02431

Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests

R. K. Hagmann,[1,2,27] P. F. Hessburg,[1,3] S. J. Prichard,[1] N. A. Povak,[3,4] P. M. Brown,[5] P. Z. Fulé,[6] R. E. Keane,[7] E. E. Knapp,[8] J. M. Lydersen,[9] K. L. Metlen,[10] M. J. Reilly,[11] A. J. Sánchez Meador,[12] S. L. Stephens,[13] J. T. Stevens,[14] A. H. Taylor,[15] L. L. Yocom,[16] M. A. Battaglia,[17] D. J. Churchill,[18] L. D. Daniels,[19] D. A. Falk,[20,26] P. Henson,[21] J. D. Johnston,[22] M. A. Krawchuk,[22] C. R. Levine,[23] G. W. Meigs,[18] A. G. Merschel,[22] M. P. North,[24] H. D. Safford,[25] T. W. Swetnam,[26] and A. E. M. Waltz[12]

[1]*College of the Environment-SEFS, University of Washington, Seattle, Washington 98195 USA*
[2]*Applegate Forestry LLC, Corvallis, Oregon 97330 USA*
[3]*USDA-FS, Forestry Sciences Laboratory, Pacific Northwest Research Station, Wenatchee, Washington 98801 USA*
[4]*USDA-FS, Pacific Southwest Research Station, Placerville, California 95667 USA*
[5]*Rocky Mountain Tree-Ring Research, Fort Collins, Colorado 80526 USA*
[6]*School of Forestry, Northern Arizona University, Flagstaff, Arizona 86011 USA*
[7]*Missoula Fire Sciences Laboratory, USDA-FS, Rocky Mountain Research Station, Missoula, Montana 59808 USA*
[8]*USDA-FS, Pacific Southwest Research Station, Redding, California 96002 USA*
[9]*Fire and Resource Assessment Program, California Department of Forestry and Fire Protection, Sacramento, California 94244 USA*
[10]*The Nature Conservancy, Ashland, Oregon 97520 USA*
[11]*USDA-FS, Pacific Northwest Research Station, Corvallis, Oregon 97333 USA*
[12]*Ecological Restoration Institute, Northern Arizona University, Flagstaff, Arizona 86011 USA*
[13]*Department of Environmental Science, Policy, and Management, University of California–Berkeley, Berkeley, California 94720 USA*
[14]*U.S. Geological Survey, Fort Collins Science Center, New Mexico Landscapes Field Station, Santa Fe, New Mexico 87508 USA*
[15]*Department of Geography, Earth and Environmental Systems Institute, The Pennsylvania State University, University Park, Pennsylvania 16802 USA*
[16]*Department of Wildland Resources and the Ecology Center, Utah State University, Logan, Utah 84322 USA*
[17]*USDA-FS, Rocky Mountain Research Station, Fort Collins, Colorado 80526 USA*
[18]*Washington State Department of Natural Resources, Olympia, Washington 98504 USA*
[19]*Department of Forest and Conservation Sciences, University of British Columbia, Vancouver, British Columbia V6T 1Z4 Canada*
[20]*School of Natural Resources and the Environment, University of Arizona, Tucson, Arizona 85721 USA*
[21]*Oregon Fish and Wildlife Office, USDI Fish & Wildlife Service, Portland, Oregon 97232 USA*
[22]*College of Forestry, Oregon State University, Corvallis, Oregon 97333 USA*
[23]*Spatial Informatics Group, Pleasanton, California 94566 USA*
[24]*USDA-FS, Pacific Southwest Research Station, Mammoth Lakes, California 93546 USA*
[25]*USDA-FS, Pacific Southwest Region, Vallejo, California 94592 USA*
[26]*Laboratory of Tree-Ring Research, University of Arizona, Tucson, Arizona 85721 USA*

Citation: Hagmann, R. K., P. F. Hessburg, S. J. Prichard, N. A. Povak, P. M. Brown, P. Z. Fulé, R. E. Keane, E. E. Knapp, J. M. Lydersen, K. L. Metlen, M. J. Reilly, A. J. Sánchez Meador, S. L. Stephens, J. T. Stevens, A. H. Taylor, L. L. Yocom, M. A. Battaglia, D. J. Churchill, L. D. Daniels, D. A. Falk, P. Henson, J. D. Johnston, M. A. Krawchuk, C. R. Levine, G. W. Meigs, A. G. Merschel, M. P. North, H. D. Safford, T. W. Swetnam, and A. E. M. Waltz. 2021. Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests. Ecological Applications 31(8):e02431. 10.1002/eap.2431

Abstract. Implementation of wildfire- and climate-adaptation strategies in seasonally dry forests of western North America is impeded by numerous constraints and uncertainties. After more than a century of resource and land use change, some question the need for proactive management, particularly given novel social, ecological, and climatic conditions. To address this question, we first provide a framework for assessing changes in landscape conditions and fire regimes. Using this framework, we then evaluate evidence of change in contemporary conditions relative to those maintained by active fire regimes, i.e., those uninterrupted by a century or more of human-induced fire exclusion. The cumulative results of more than a century of research document a persistent and substantial fire deficit and widespread alterations to

Manuscript received 17 November 2020; revised 4 March 2021; accepted 22 March 2021; final version received 8 July 2021. Corresponding Editor: David S. Schimel.
[27] E-mail: hokulea@uw.edu

ecological structures and functions. These changes are not necessarily apparent at all spatial scales or in all dimensions of fire regimes and forest and nonforest conditions. Nonetheless, loss of the once abundant influence of low- and moderate-severity fires suggests that even the least fire-prone ecosystems may be affected by alteration of the surrounding landscape and, consequently, ecosystem functions. Vegetation spatial patterns in fire-excluded forested landscapes no longer reflect the heterogeneity maintained by interacting fires of active fire regimes. Live and dead vegetation (surface and canopy fuels) is generally more abundant and continuous than before European colonization. As a result, current conditions are more vulnerable to the direct and indirect effects of seasonal and episodic increases in drought and fire, especially under a rapidly warming climate. Long-term fire exclusion and contemporaneous social-ecological influences continue to extensively modify seasonally dry forested landscapes. Management that realigns or adapts fire-excluded conditions to seasonal and episodic increases in drought and fire can moderate ecosystem transitions as forests and human communities adapt to changing climatic and disturbance regimes. As adaptation strategies are developed, evaluated, and implemented, objective scientific evaluation of ongoing research and monitoring can aid differentiation of warranted and unwarranted uncertainties.

Key words: climate adaptation; Climate Change and Western Wildfires; ecosystem management; fire exclusion; forested landscapes; frequent fire; high-severity fire; landscape restoration; multi-dimensional fire regimes; multi-scale spatial patterns; reference conditions; wildfire adaptation.

Introduction

Social and ecological impacts of large and intense wildfires present enormous challenges to land and resource managers of western North America (Franklin and Agee 2003, North et al. 2015, Moreira et al. 2020, Hessburg et al. 2021). In the near term, wildfire frequency, area burned, and area burned at high severity will likely continue to increase as the climate warms; however, despite recent climatically driven increases in area burned, fire deficits in seasonally dry forests remain high (reviewed by Hessburg et al. 2021). After more than a century of fire exclusion (Fig. 1), increased density, abundance, and continuity of live and dead vegetation interact with increased seasonal warming and drying to drive wildfire severity (Miller et al. 2009*b*, Steel et al. 2015, Parks et al. 2018, Parks and Abatzoglou 2020). While modern wildfire management suppresses most fire starts, those that exceed suppression capacity account for the majority of burned area, often during the most extreme fire weather (North et al. 2015, Moreira et al. 2020). A paradigm shift that recognizes wildfire and extreme fire weather as inevitable and characteristic of seasonally dry forested ecosystems may better foster fire- and climate-adapted forests and human communities (Moreira et al. 2020).

Some restoration of low- and moderate-severity fire is occurring (Parks et al. 2014, Stevens-Rumann et al. 2016, Walker et al. 2018*a*, Brown et al. 2019, Kane et al. 2019, Mueller et al. 2020). However, as described above, current live and dead fuel loads and management emphases diminish the likelihood of recapturing the once extensive influence of low- and moderate-severity fires. Departures from the successional patterns that resulted from and supported active fire regimes (*i.e.*, those uninterrupted by more than a century of human-induced fire exclusion) have left many forests vulnerable to the direct and indirect effects of seasonal and episodic increases in drought and fire, especially under a warming climate (Allen et al. 2002, Noss et al. 2006, Daniels et al. 2011, Chavardès et al. 2018, Keane et al. 2018, Stephens et al. 2018*a*, Bryant et al. 2019).

Fire regime changes also influence other pattern-process interactions and ecosystem functions, including primary productivity relations, carbon and nutrient cycling, evapotranspiration and distributed hydrology, and the movement and persistence of organisms (Turner 1989, Bowman et al. 2009). Thus, implementation of scientifically credible adaptation strategies can benefit numerous social values, including quantity and quality of water supply, stability of carbon stores, and air quality (Stephens et al. 2020) as well as Indigenous fire stewardship and food security (Lake and Long 2014, Norgaard 2014, David et al. 2018, Sowerwine et al. 2019).

Proactive management informed by historical and contemporary forest and fire ecology can strengthen resistance to disturbance and better align forest ecosystems with rapidly changing climatic and disturbance regimes (reviewed by Hessburg et al. 2021, Prichard et al. 2021). Reducing the abundance and connectivity of fuels that accumulated over more than a century of fire exclusion can moderate ecosystem transitions and provide numerous ecological and socioeconomic benefits (reviewed by Prichard et al. 2021). Indigenous fire stewardship practices can inform active management that achieves shared values to benefit tribes, local communities, and the broader society when tribes contribute to leadership and management of collaborative restoration partnerships (Lake et al. 2018, Long and Lake 2018, Long et al. 2020).

Implementing adaptation strategies at scales sufficient to alter contemporary disturbance regimes and recover other ecosystem functions associated with widespread low- and moderate-severity fires involves significantly increasing active management of forested landscapes (Spies et al. 2006, North et al. 2015, Stephens et al. 2016, Barros et al. 2017). Uncertainty, trade-offs, and risks are inevitable components of both action and

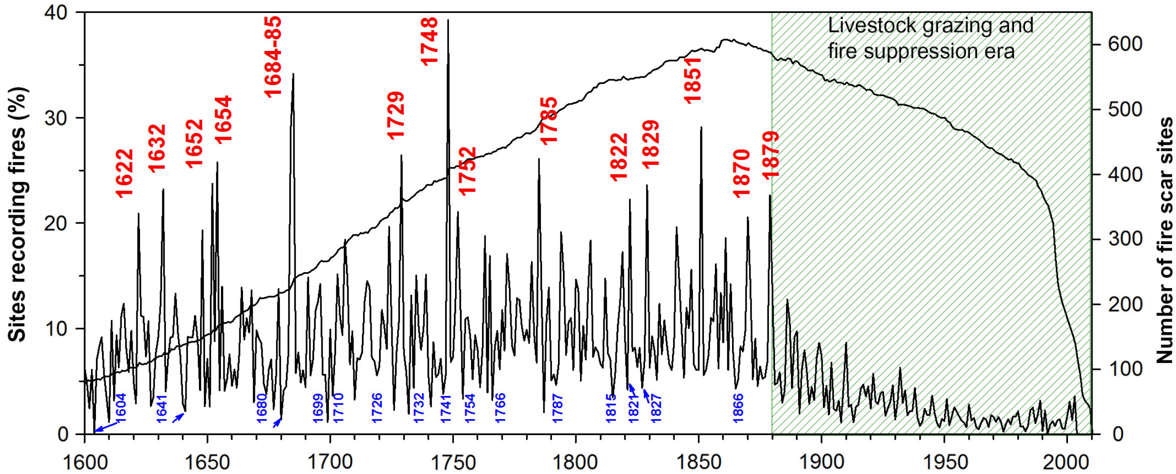


Fig. 1. Across western North America, fire frequency decreased substantially following expansion of colonization by Europeans, intensive livestock grazing, decimation of Indigenous populations and suppression of Indigenous burning in the late 19th century. The combined record of fire occurrence from more than 800 forest and woodland sites, the largest network of tree-ring-based fire-scar chronologies in the world, illustrates this regionwide decrease in fire frequency. Reprinted from Swetnam et al. (2016) with the author's permission.

inaction, whether proactive or reactive; ongoing research, multi-party monitoring, and adaptive management seek to address these components and build trust in proactive management (reviewed by Hessburg et al. 2021, Prichard et al. 2021). While integral to the development of knowledge, dissent in the scientific literature can contribute to conflict, confusion, and lack of consensus in stakeholders, e.g., environmental and conservation organizations and the general public (Maier and Abrams 2018). When fostered by incomplete assessment of the best available science (Esch et al. 2018), this lack of consensus may unnecessarily delay development and implementation of constructive new solutions and policies (reviewed by Hessburg et al. 2021).

To aid those engaged in designing, evaluating, and implementing science-based adaptation options, we evaluate lingering uncertainties about the high-severity component of historical and contemporary fire regimes (e.g., see Moritz et al. 2018). We first provide a framework for objectively assessing change in the structure, composition, and fire regimes of seasonally dry, fire-excluded forested landscapes. We then review key aspects of more than a century of research and observations of changes in forest conditions and fire regimes and the influence of those changes on contemporary processes and functions.

We contrast the evidence of change with evidence suggesting that management that reduces forest density to mitigate high-severity disturbance lacks sound ecological support. Over the past two decades, the ecological and policy implications of these publications (e.g., Baker and Ehle 2001, Williams and Baker 2012, DellaSala and Hanson 2019) have garnered substantial attention and fostered confusion about the best available science. To aid evaluation of the relative merit of this body of evidence and counter-evidence to contemporary management, we also synthesize independent, peer-reviewed evaluations of methodologies used in the counter-evidence publications.

Framework For Evaluating Change

Terms of reference

Forest types.—We focus on temperate forests of interior western North America (Fig. 2). This biogeoclimatically diverse region supports a wide range of forest types composed of broadleaf and coniferous species. Dominant species on the dry end of the gradient include ponderosa and Jeffrey pine (*Pinus ponderosa* and *P. jeffreyi*) and some oak species (*Quercus* spp.). As moisture increases or fire frequency decreases, species with higher shade tolerance and lower drought and fire tolerance increasingly dominate; these include Douglas-fir (*Pseudotsuga menziesii*); western larch (*Larix occidentalis*); sugar, western white, and southwestern white pine (*Pinus lambertiana*, *P. monticola*, and *P. strobiformis*); incense-cedar (*Calocedrus decurrens*); and grand and white fir (*Abies grandis* and *A. concolor*). As mean annual temperatures decrease with elevation or cold air drainage, forests are increasingly dominated by lodgepole pine (*Pinus contorta*); aspen (*Populus tremuloides*); red, silver, and subalpine fir (*Abies magnifica*, *A. amabilis*, and *A. lasiocarpa*); mountain hemlock (*Tsuga mertensiana*); Engelmann spruce (*Picea engelmannii*); or whitebark pine (*Pinus albicaulis*).

Using Landfire (Rollins 2009) Biophysical Settings, we classify these forest types as either cold, moist, or dry (Fig. 2, Appendix S1). We exclude rainforests, coastal forests, and Douglas-fir–western hemlock (*Tsuga heterophylla*) forests of the Coast Ranges and the west slope of the Cascade Mountain Range. These mesic and coastal

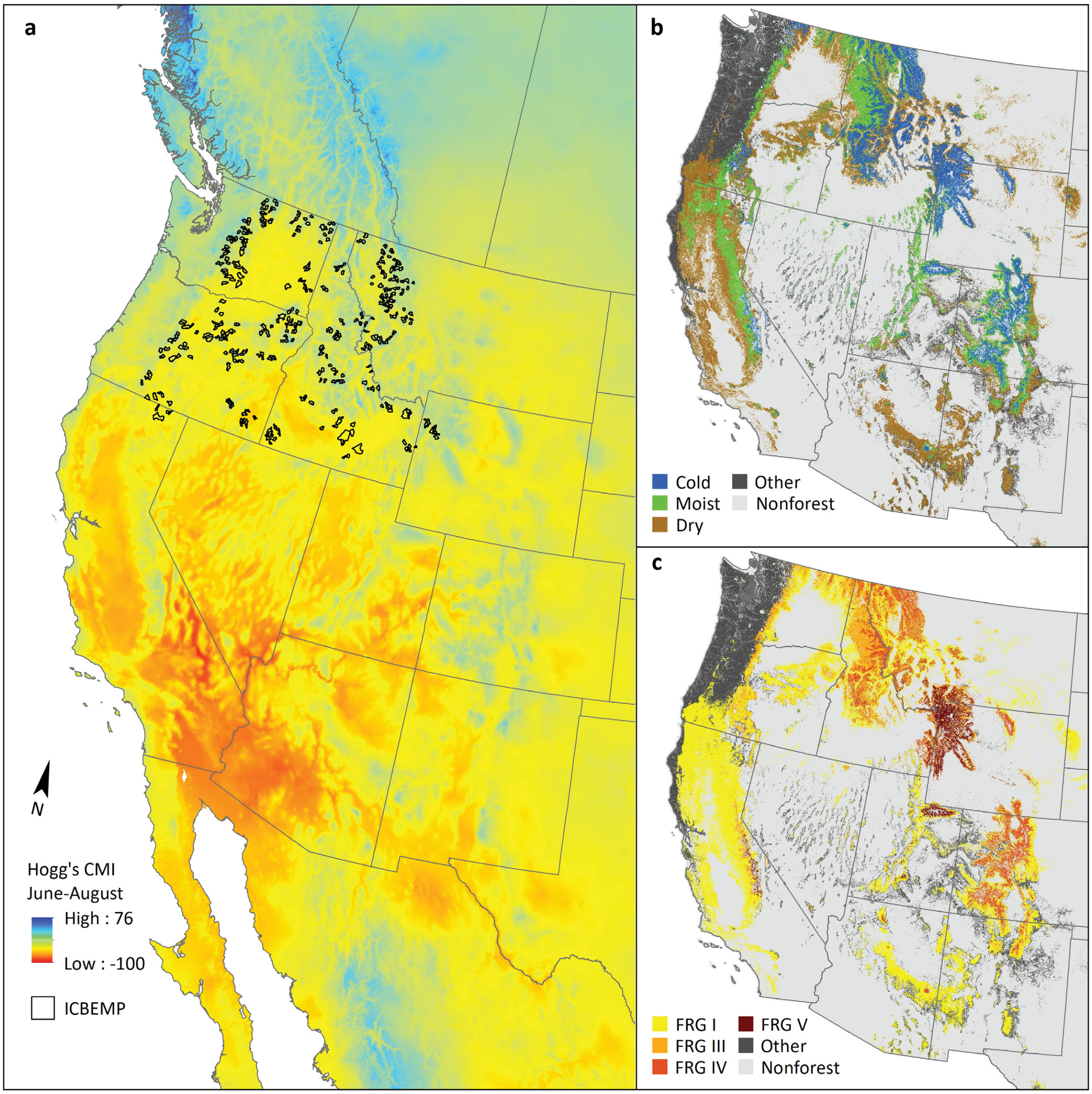


Fig. 2. (a) Summer available moisture and Interior Columbia Basin Ecosystem Management Project (ICBEMP) sampled area, (b) cold, moist, and dry forest types, and (c) fire regime group (FRG) classes. FRG classes reflect strong regional variation in biogeoclimatic conditions between northern and southern North America generally and between the Rocky Mountain ecoregions and those dominated by lower elevations. FRG I, fire return interval ≤ 35 yr, low and mixed severity; FRG III, fire return interval 35–200 yr, low and mixed severity; FRG IV, fire return interval 35–200 yr, replacement or high-severity; FRG V, fire return interval > 200 yr, any severity. Portions of the study area that extend into Mexico and Canada are not shown in b and c because Landfire data are not available for these regions. Data sources are (a) Hogg's Climate Moisture Index (Hogg 1997) from ClimateWNA (Hamann et al. 2013, climatewna.com); (b,c) Landfire (Rollins 2009, landfire.gov).

forests are typically associated with infrequent high-severity fire; however, a growing body of research suggests that low- to moderate-severity fire also likely affected their resistance and resilience (Daniels and Gray 2006), particularly in drier portions of their range (Spies et al. 2018*b*) and where Indigenous people commonly burned the forest (Pellatt and Gedalof 2014, Hoffman et al. 2017, 2019). Resilience is the capacity of an ecosystem to recover its essential characteristics (including taxonomic composition, structure, ecosystem function, and process rates) following a disturbance, whereas resistance is the property of an ecosystem to remain essentially unchanged when disturbed (Grimm and Wissel 1997). Additionally, forest types dominated by Douglas-fir, western hemlock, and western redcedar (*Thuja occidentalis*) do occur in the interior east of the

Cascade crest, often mixed with the mesic or cold forest species listed above.

Reference conditions.—The concept of "departure" necessitates knowledge of past conditions and their variability, often referred to as "reference conditions" or the historical or natural "range of variation" (Morgan et al. 1994, Hessburg et al. 1999*b*, Swetnam et al. 1999, Keane et al. 2009). Comparison of contemporary conditions with reference conditions provides insight into the magnitude, rate, and direction of change (Higgs et al. 2014). Timing of fire exclusion (Fig. 1) varied widely, but commonly accompanied disruption of Indigenous burning and expansion of unregulated grazing of livestock by European settlers, often many decades to more than a century prior to mechanized fire suppression, logging, and land development (Marlon et al. 2012, Swetnam et al. 2016).

Reference baselines are commonly constrained to two to four centuries prior to widespread colonization by Europeans (ca. 1850). Climate and potential vegetation patterns in this period were broadly similar to those of the early 20th century, and data sources with high temporal resolution, *e.g*., tree-rings and fire scars, can be used to reconstruct environmental conditions for this period (Morgan et al. 1994, Falk et al. 2011). Palaeoecological and archaeological evidence provide insight into the influence of climate variation (Betancourt et al. 1990, Whitlock and Bartlein 1997, Beaty and Taylor 2009, Marlon et al. 2012, Swetnam et al. 2016, Bigio et al. 2017) and Indigenous resource and fire use (Kaye and Swetnam 1999, Klimaszewski-Patterson and Mensing 2016, Roos et al. 2021) as drivers of change over longer time frames. Areas with relatively intact forest conditions or fire regimes (i.e., active fire regimes) provide insight into how historical forests and landscapes might have operated under contemporary climate and disturbance regimes (Stephens and Fulé 2005, Collins et al. 2009). Evaluation of landscape-level forest structure and composition with high spatial resolution, however, relies more heavily on conditions that existed in the early to mid-20th century, the timeframe of the earliest available aerial and oblique photos (Hessburg et al. 2000). Since no single approach addresses all relevant scales of observation, multiple lines of independent corroborating evidence are needed to quantify spatial and temporal variation in reference conditions.

Multi-scale, multi-proxy records increase inference space

While individual methods are particularly well suited for evaluating aspects of forest conditions and disturbance regimes at specific temporal and spatial scales (Wiens et al. 2012, Morgan et al. 2014, Yocom Kent 2014), multi-proxy studies can compensate for limitations in each data source (Swetnam et al. 1999, Daniels et al. 2017). Incorporating several lines of evidence (e.g., multi-scale and multi-proxy studies, meta-analyses, or simulation models) can increase confidence in results, broaden inference space, clarify the existence and extent of change, and provide insight into change mechanisms (Whitlock et al. 2004, Taylor et al. 2016).

Multi-proxy, multi-scale research also reveals that, when considered in isolation, lack of evidence of change at any single scale of observation or in any single sampled attribute may mislead interpretation of the degree of ecosystem departures. For example, studies conducted at plot or patch-scales may fail to capture variability of vegetation conditions and fire severity across larger landscapes (Marcoux et al. 2015). Thus, while change in one or more aspect of a fire regime, e.g., percentage of land affected by high-severity fire, may have occurred, it may not be evident at all scales of observation. Similarly, while the percentage of the land area affected by high-severity fire may not have changed, spatial patterns of high-severity fire may have (Collins et al. 2017). Reliance on any one methodology or scale of observation is insufficient to understanding the scope of changes given the multi-scale complexities of climate–vegetation-disturbance feedbacks and their influence on patterns and processes (Falk et al. 2019).

Forest conditions exist at multiple spatial scales.—Spatial patterns of vegetation reflect strong linkages between biogeoclimatic conditions, disturbance and succession processes, and plant physiology that vary over space and time. Here, we consider the dominant factors operating at three spatial scales (Fig. 3): broad (>10,000 ha), meso (100 to 10,000 ha), and fine (<100 ha). Each scale of observation is important to understanding vegetation change, subsequent interactions with disturbance processes (i.e., fire, drought, insects, and pathogens), and potential future conditions (Keane et al. 2009, Wiens et al. 2012, Hessburg et al. 2019). To assess whether forest vegetation conditions are trending away from a given baseline, it is essential to consider changes at several spatial scales and in cross-scale linkages (sensu Wu and Loucks 1995).

At broad scales, forests exist within a patchwork of nonforest physiognomic types, including herbland/grassland, shrubland, woodland or savannah, and bare ground. Physiognomic types generally reflect the range of temperature, precipitation, solar radiation, soil, and geomorphic conditions to which they are best adapted. However, overlapping disturbances occurring in rapid succession, e.g., frequent fire, can override site potential, leading to relatively stationary patches of nonforest on forest-capable sites (Coppoletta et al. 2016, Prichard et al. 2017, Coop et al. 2020, McCord et al. 2020).

At meso-scales, heterogeneous patterns of forest and nonforest structures and compositions reflect the history of interacting and overlapping disturbances combined with succession and stand dynamics processes (Perry et al. 2011, Hessburg et al. 2016, 2019) as well as biogeoclimatic conditions, e.g., soil types (Winthers et al. 2005). The result is a mosaic of forest successional patches that reside within the larger physiognomic

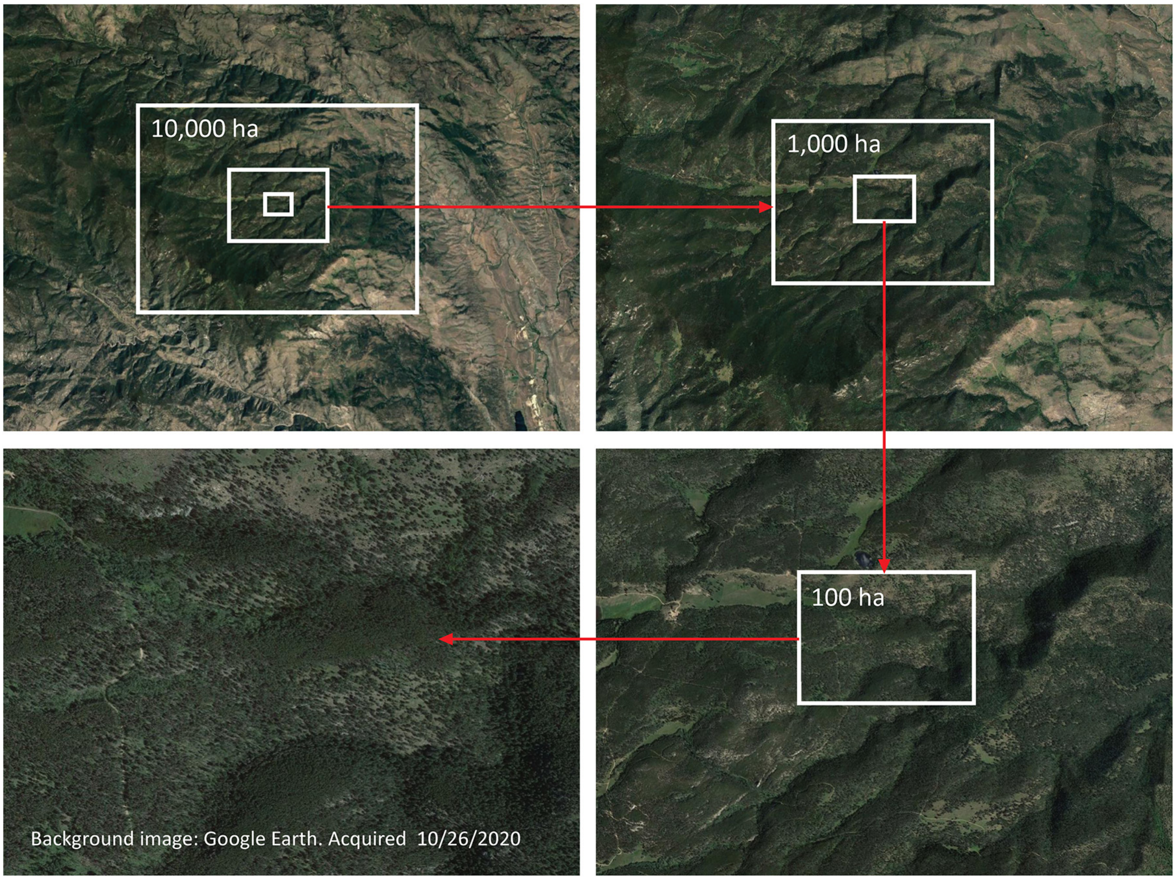


FIG. 3. At broad (>10,000 ha), meso (100 to 10,000 ha), and fine (<100 ha) scales, spatial patterns of vegetation are influenced by biogeoclimatic conditions, disturbance and succession processes, and plant physiology. Heterogeneity is evident at each spatial scale and can influence the spread of disturbances (e.g., fire) and the movement of resources (e.g., water and sediment) as well as species. Area shown is west of Fort Collins, Colorado, USA.

patchwork. As with physiognomic types, frequent disturbance can override site potential and inhibit succession to closed-canopy forests or dominance by fire-intolerant species (Agee 1996, 1998, Hessburg et al. 2005, North et al. 2009, Stine et al. 2014).

At fine scales, physiological and anatomical traits of tree, shrub, and herb species and interactions with soils influence community structure and composition, canopy and gap dynamics, variation in fuel load, and fire severity (North et al. 2002, Meyer et al. 2007, Reynolds et al. 2013, Strahan et al. 2016, Laughlin et al. 2017, Stevens et al. 2020). These include traits that determine interactions with fire for individual trees (e.g., bark thickness and needle shape) and populations (e.g., reproduction and germination strategies). Overlapping disturbances also modify the imprint of previous events at fine spatial scales (Hansen et al. 1991, Franklin and Van Pelt 2004). Thus, in forests that burned frequently, variation in successional stages typically occurred at very fine spatial scales (<1 ha) resulting in a mosaic of individual trees, clumps of trees, and openings, rather than patches or stands (Franklin and Van Pelt 2004, Kaufmann et al. 2007, Larson and Churchill 2012, Churchill et al. 2013, 2017, Lydersen et al. 2013, Fry et al. 2014, Ng et al. 2020).

Fire regimes are multi-dimensional.—Multiple dimensions of individual fires (Hessburg et al. 2021: Table 1) interacting in a relatively persistent pattern over long periods of time collectively comprise a holistic notion of a fire regime (Agee 1996, Sugihara et al. 2018). Fire frequency and severity are major drivers of ecological and evolutionary response (Keeley 2012). However, limiting definitions of fire regimes to the frequency and severity that dominate a given area (e.g., frequent low-severity or infrequent high-severity) oversimplifies ecological understanding of wildfire regimes, and impedes detection of departures and projection of future conditions (Brown et al. 2008, Collins et al. 2017). Multiple other aspects of fire regimes (e.g., area burned, seasonality, spatial complexity) must also be considered to understand the natural variability of active fire regimes and evaluate departures (Daniels et al. 2017).

Fire severity is often measured as the percentage mortality of tree biomass (e.g., tree basal area or canopy cover) after each fire event. Conventional definitions of

Table 1. A sample of the regional syntheses and meta-analyses providing multi-proxy, multi-scale assessments of historical and contemporary forest and fire ecology.

Region and description	Citations
Western North America	
More than 800 fire-scar studies documented abrupt decline in fire frequency in the late 19th century and provide ecological insights into variation in top-down and bottom-up drivers of historical fire regimes.	Falk et al. (2011), Swetnam et al. (2016), Daniels et al. (2017)
Substantial departures in contemporary fire regimes and live and dead vegetation patterns across dry, moist, and cold forested landscapes increase vulnerability of forest ecosystems to drought and fire.	Hessburg et al. (2019)
Canada	
Development and paradigm shift in wildland fire research over past 50 yr.	Coogan et al. (2020)
Climate change impacts on fire regimes and impacts of contemporary fire regimes on social and ecological systems.	Coogan et al. (2019)
Western United States	
Variation in fire activity over the past 3,000 yr.	Marlon et al. (2012)
Fire deficit relative to area expected to burn without fire suppression given contemporary climate 1984–2012; area burned and fire severity increased 1985–2017.	Parks et al. (2015), Parks and Abatzoglou (2020)
Influence of traditional tribal perspectives on ecosystem restoration.	Long et al. (2020), Roos et al. (2021)
Correspondence between conifer species traits conferring fire resistance and independent assessments of historical fire regimes.	Stevens et al. (2020)
Human influence on contemporary fire regimes.	Balch et al. (2017)
Evaluation of conifer regeneration up to 69 yr post fire.	Stevens-Rumann and Morgan (2019)
Colorado and Wyoming Front Ranges	
Historical and contemporary ecology of ponderosa pine and dry mixed-conifer forests.	Addington et al. (2018)
Fire regimes in ponderosa pine forests.	McKinney (2019)
Historical and contemporary ecology of selected national forests.	Dillon et al. (2005), Meyer et al. (2005*a*, *b*), Veblen and Donnegan (2005)
Southwestern United States	
Historical and contemporary ecology of ponderosa pine and dry mixed-conifer forests and forest–grassland landscape complexes.	Reynolds et al. (2013), Dewar et al. (2021)
Sierra Nevada bioregion of California	
Historical and contemporary ecology of ponderosa and Jeffrey pine and mixed-conifer forests.	SNEP (1996), North et al. (2009, 2016), Safford and Stevens (2017), van Wagtendonk et al. (2018*a*)
Historical and contemporary ecology of red fir and subalpine forest types.	Meyer and North (2019), Coppoletta et al. (2021)
Northeastern California plateaus	
Historical and contemporary ecology of dry conifer forests.	Riegel et al. (2018), Dumroese and Moser (2020)
Northern California	
Historical and contemporary ecology of forested landscapes.	Skinner et al. (2018), Stephens et al. (2018*b*, 2019), Bohlman et al. (2021)
Pacific Northwest	
Departures in contemporary fire regimes.	Reilly et al. (2017), Metlen et al. (2018), Haugo et al. (2019)
Historical and contemporary ecology of ponderosa pine forests in Oregon and Washington; vulnerability of contemporary forests and expanding wildland urban interface to increasing drought and fire severity.	Merschel et al. (2021)
Historical and contemporary ecology of moist mixed conifer forests in seasonally dry landscapes in Oregon, Washington, and Northern California.	Perry et al. (2011), Spies et al. (2018*b*, 2019), Stine et al. (2014), Hessburg et al. (2016)
Columbia River Basin in northwestern United States	
The Interior Columbia Basin Ecosystem Management Project (ICBEMP) used standard aerial photogrammetric methods, repeat photo-interpretation, and a quantitatively representative sampling scheme to build a data set of wall-to-wall, meso-scale landscape reconstructions for 337 watersheds, mean area 9,500 ha. ICBEMP also incorporated broad-scale succession and disturbance simulation modeling calibrated with the meso-scale results.	Lehmkuhl et al. (1994), Huff et al. (1995), Hann et al. (1997), Hessburg et al. (1999, 2000, 2005), Wisdom (2000). Raphael et al. (2001), Hessburg and Agee (2003)

fire regimes (e.g., Agee 1996) generally reflect the cumulative abundance of low- (<20%), moderate- (20–70%), and high- (>70%) severity fire in individual fire events at broad temporal and spatial scales. However, each of these severity classes (as well as other commonly used terms like mixed or variable severity), encompass a wide range of potential ecological outcomes, i.e., the difference between outcomes at either end of the severity gradient in each of these classes can be substantive. Additionally, these severity classes do not consider spatial patterns of fire severity; without them, however, assessment of the ecological impacts of fire events is incomplete (Miller and Quayle 2015, Collins et al. 2017, Shive et al. 2018, Walker et al. 2019).

Evaluating Evidence of Change

The cumulative results of more than a century of research and observation from numerous disciplines document regional and subregional variation in historical and contemporary forest and fire ecology (Table 1). Here, we focus on key elements from this vast body of work to illustrate the magnitude of change in forested landscapes. Comprehensive reviews of departures within and among forest types and regions are available in existing syntheses, meta-analyses, and regionwide studies (Table 1).

We begin with a landscape evaluation of change in vegetation spatial patterns and fire regimes across 61 million ha (Hann et al. 1997, 1998) that encompass the highest concentration of cold and moist forest in the interior western United States (Fig. 2). Landscape assessments that evaluate a broad variety of attributes of fire regimes and forest conditions can reduce the risk of oversimplifying or misrepresenting spatiotemporal variability in fire severity and forest conditions. The substantial departures documented in this assessment underscore those documented in numerous other studies both within this region and in predominantly warmer, drier ecoregions (Table 1). We also consider changes in extent of nonforest, which can reflect significant changes in disturbance processes over space and time (Perry et al. 2011, Hessburg et al. 2016, 2019, Coop et al. 2020).

Next, we review evaluations of departures from active fire regimes. As physical evidence of fire occurrence, fire scar records remain a primary means of exploring historical fire ecology. Networks of fire-scar studies emerging from the cumulative results of a century of tree-ring studies enable insights into landscape and climate controls on fire (Falk et al. 2011). Along with novel research designs for evaluating dendrochronological records of fire history (Farris et al. 2010, Tepley and Veblen 2015, Greene and Daniels 2017, Naficy 2017), landscape-level assessments and simulation models encompassing multiple forest types can address concerns that sampling bias of fire-scar studies favors detection of low-severity fire regimes (e.g., see arguments in Hessburg et al. 2007).

Throughout, we reference results of landscape succession and disturbance models, which provide an important means of extrapolating geographically limited historical data across large areas, over long time periods, under diverse climatic conditions, and over a wide range of fuel characteristics (Bradstock et al. 1998, Keane et al. 2004, Barros et al. 2017). Simulation modeling allows ecologists to integrate what is currently known to evaluate hypotheses that enhance our collective understanding of fire and its distributed effects (Spies et al. 2017, Barros et al. 2018, Keane 2019). Landscape succession and disturbance models combine fire history and biotic information about forest species as parameters (Keane 2019, Loehman et al. 2020) to inform simulations of past, present, and future landscape-wildfire dynamics (Keane et al. 2004, He et al. 2008). Perhaps most importantly, these models can inform and evaluate management scenarios; they can be used to simulate multiple future climate, management, and exotic species scenarios that can then be compared with simulated historical conditions under a consistent framework to evaluate risks, trade-offs, and uncertainties (Keane 2019).

Forest and nonforest conditions are significantly departed

The Interior Columbia Basin Ecosystem Management Project (hereafter, ICBEMP) documented widespread forest expansion and densification between early (primarily 1930s–1950s) and late (primarily 1990s) 20th century (Hann et al. 1997, 1998, Hessburg et al. 2000, 2005). The ICBEMP encompassed the range of interior forest environments distributed across Washington, Idaho, Montana, Oregon, and northern California. Using repeat photo-interpretation, standard aerial photogrammetric methods, and a quantitatively representative sample (337 watersheds, mean area ~10,000 ha), the ICBEMP meso-scale assessment (Hessburg et al. 1999*a*, 2000, 2005, Hessburg and Agee 2003) evaluated change in forest landscape patterns across the 20th century, and the effects of those changes on fuel and fire regime conditions. The results of this meso-scale assessment were used to calibrate broadscale simulations of changes across the entire ICBEMP area (Keane 1996, Hann et al. 1997).

Both assessments (repeat photo-interpretation and simulation modeling) found that high-severity disturbances at lower frequencies and low- and moderate-severity disturbances at higher frequencies collectively reduced total forested area and perpetuated relatively widespread herbland/grassland, shrubland, woodland, and, often, open-canopy forest, which tended to support high fire spread rates, low flame lengths, and low fireline intensities under most fire weather conditions (Keane 1996, Hann et al. 1997, Hessburg et al. 2016, 2019). By the late 20th century, dry, moist, and cold forest landscapes had become more densely forested, resulting in homogenization of previously diverse forest and nonforest successional conditions, elevated

vulnerability to contagious disturbances, and loss of key habitats (Wisdom 2000, Raphael et al. 2001). These changes were apparent despite extensive logging in the mid to late 20th century and impacts of fire exclusion evident by the 1930s in some areas. By the late 20th century, the area likely to support fire regimes of low-severity had been reduced by 53%, mixed-severity remained roughly the same (although it shifted to sites that supported low-severity fire regimes prior to fire exclusion), and high-severity had nearly doubled (Fig. 4, Keane 1996).

In studies spanning western North America, the extensive influence of frequent low- and moderate-severity fires in maintaining open-canopy dry forests and woodlands has been repeatedly documented (Table 1). Although not as prevalent, departures associated with the loss of low- to moderate-severity fire are also documented in moist and cold forests. Examples include lodgepole pine in the foothills of the Rocky Mountains in Alberta (Amoroso et al. 2011) and in cold-air drainages in the central Oregon Pumice Plateau ecoregion (Heyerdahl et al. 2014, Hagmann et al. 2019); mixed-conifer and subalpine forests in the Canadian Cordillera (Marcoux et al. 2015, Chavardès and Daniels 2016, Rogeau et al. 2016) and southwestern United States (Margolis and Malevich 2016, Johnson and Margolis 2019); red fir forests in California's Sierra Nevada ecoregion (Meyer et al. 2019), and the ICBEMP study area (Fig. 2) described above. As in the ICBEMP area (Fig. 4), increased surface fuel loads and canopy connectivity in mid-elevation forests likely influence the frequency of crown fire spread into more mesic high-elevation forests in the southwest as well (O'Connor et al. 2014*a*).

Oblique and aerial imagery from the early 20th century document abundant nonforest cover in dry, moist, and cold forest landscapes. The William Osborne survey of Oregon and Washington in the 1930-1940s (Fig. 5) encompasses nearly 1,000 panoramas (120°) taken on ridgetops and at fire lookouts, and the Geological Survey of Canada systematically collected approximately 120,000 high-resolution oblique images from 1880 to 1950 across the mountains of western Canada (Higgs et al. 2009; photos *available online*).[28] As in the ICBEMP assessment, repeat photography from other regions shows substantial landscape change through expansion and densification of forest and consequent reduction in open-canopy forest and nonforest. Examples include high-elevation ecosystems in the Pecos Wilderness, New Mexico (deBuys and Allen 2015); pine and mixed-conifer forest over 100,000 ha in northern Sierra Nevada, California (Lydersen and Collins 2018); ponderosa pine in the Black Hills, South Dakota (Grafe and Horsted 2002) and Colorado Front Range (Fig. 6; Veblen and Lorenz 1991); and widespread change across elevations in the Canadian Rocky Mountains (Rhemtulla et al. 2011, Fortin et al. 2019, Stockdale et al. 2019*a*, Trant et al. 2020).

From broad- to fine-scales (Fig. 3), the nonforest patchwork influences landscape resilience and fire delivery to adjacent forest types. Flashy fuels, such as graminoids in grasslands, open-canopy forests, and sparse woodlands, may readily spread fire to adjacent cover types (Gartner et al. 2012, Conver et al. 2018, Prichard et al. 2018). Moreover, flashy fuels are typically the first to recover moisture content in the hours after sunset, making them important to restricting the diurnal flow of some wildfires (Simpson et al. 2016). Fine-scale treeless openings, highly variable in shape and abundance (Figs. 5, 6), provided numerous functions, including nutrient cycling and fostering biodiversity, in addition to influencing the delivery of fire to adjacent areas (North et al. 2005*b*, Larson and Churchill 2012, Churchill et al. 2017, Matonis and Binkley 2018, LeFevre et al. 2020). Changes to spatial patterns of landscape and forest structure (Figs. 5, 6) also influence aspects of the hydrologic cycle (e.g., evapotranspiration, soil water dynamics, snow interception, snow water equivalent, and snow melt timing), which can substantially reduce water available to downstream ecosystems (Boisramé et al. 2017*b*, 2019, Schneider et al. 2019, Singer et al. 2019, Ma et al. 2020, Rakhmatulina et al. 2021).

Multiple factors, including fire exclusion, have contributed to a reduction of nonforest cover and expansion of dry, moist, and cold closed-canopy forest since the early 19th century (Hessburg and Agee 2003, Chavardès et al. 2018, Eisenberg et al. 2019, Hessburg et al. 2019, Stockdale et al. 2019*a*). While the departures described above may not be evident in all sampled areas or at all spatial scales, the preponderance of evidence demonstrates that the landscape surrounding apparently unchanged ecosystems has very likely changed even if a particular patch has not. In other words, fuel loads and continuity may be higher than historical levels for a landscape although not necessarily for all patches in that landscape.

Fire regimes are significantly departed

One of the key findings to emerge from nearly every tree-ring reconstruction of fire history is a widespread reduction in fire frequency in the 20th century (Fig. 1) compared to preceding centuries (Falk et al. 2011, Marlon et al. 2012, Swetnam et al. 2016, Coogan et al. 2020). Paired tree-ring and sedimentary charcoal-based fire histories from the same locations show 20th-century decreases in fire occurrence that are unprecedented in recent millennia (Allen et al. 2008, Beaty and Taylor 2009, Swetnam et al. 2009).

Frequent fire reduces the intensities and severities of subsequent fires by maintaining tree densities and live and dead fuel loads at levels below those that local site productivity could readily support (Reynolds et al. 2013, Stine et al. 2014, Safford and Stevens 2017, Addington

[28] maps.tnc.org/osbornephotos/ and iamwho.com/cdv2/pages/byname.htm

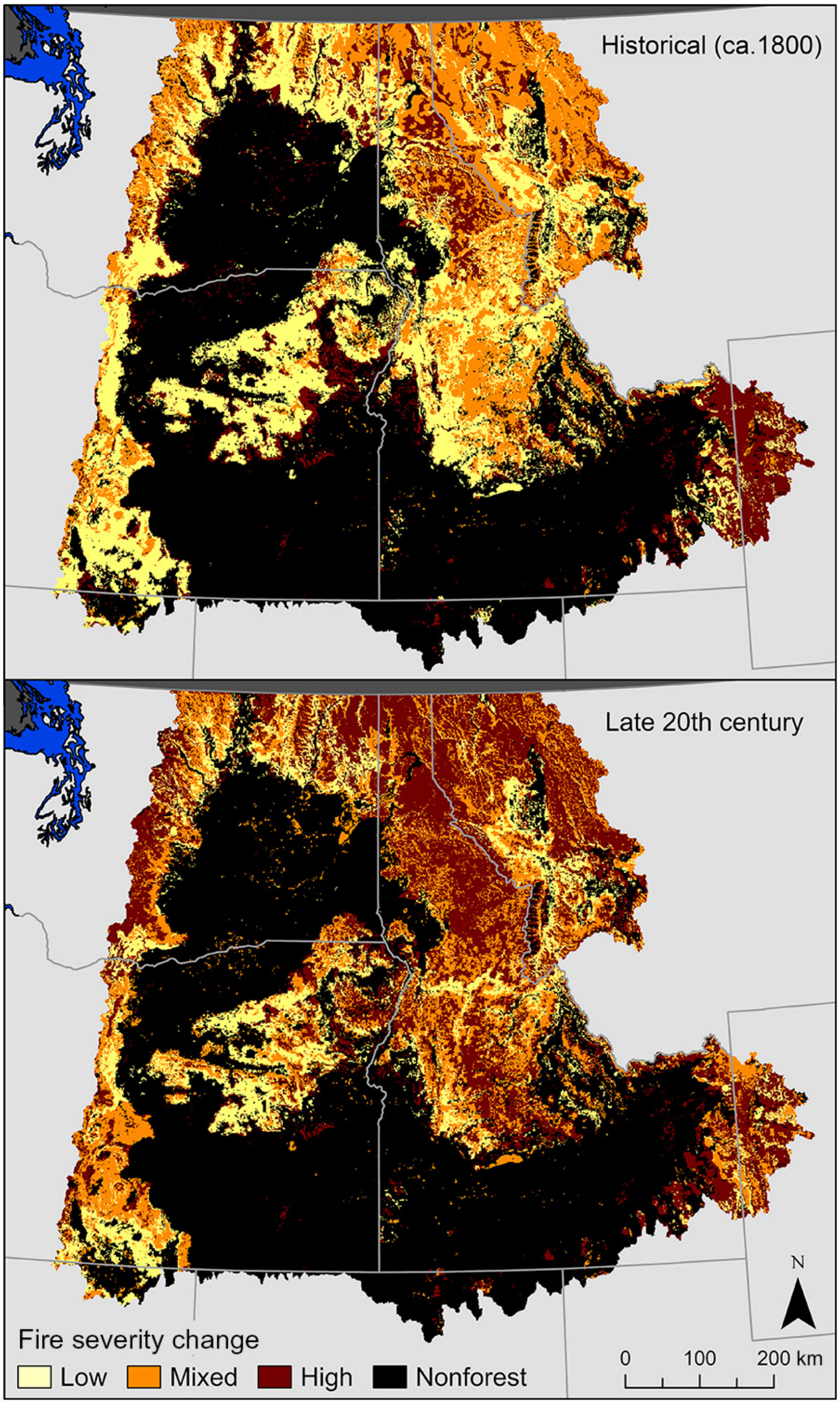


Fig. 4. Broadscale (1-km^2 pixel) map of transitions from historical (ca. 1800) to late 20th century fire-severity classes in the Interior Columbia Basin. Adapted from Hessburg et al. (2005).

et al. 2018, Battaglia et al. 2018). Overlapping fires limited the spread of crown fire and other contagious processes (e.g., insect outbreaks and disease epidemics, Hessburg et al. 1994, 1999*b*) by reinforcing discontinuities in canopy cover, species composition, tree size and age classes, and surface fuel abundance (Roccaforte et al. 2008, Collins et al. 2009, Fulé et al. 2012*a*, van Wagtendonk et al. 2018*b*). Absence of frequent fire provides

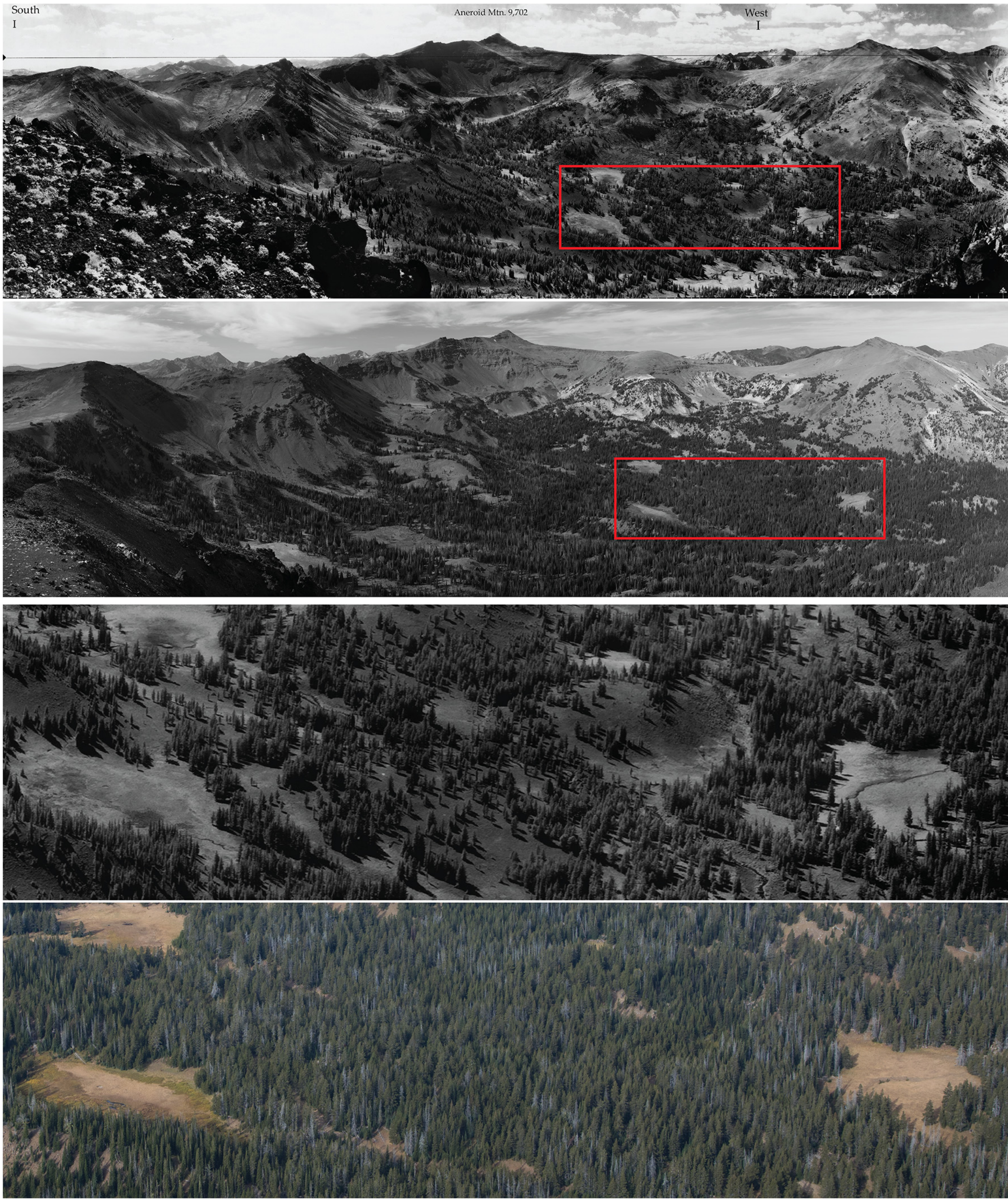


FIG. 5. Repeat photography from 1936 and 2018 demonstrates departure in spatial patterns of wet and dry meadows and cold forest successional conditions resulting from the densification and expansion of forest cover under the influence of fire exclusion, Eagle Cap Wilderness, Wallowa Mountains, Oregon. Bottom pair shows close-up of area outlined in red in the top pair. Top photo in each pair is a U.S. Forest Service 120-degree Osborne panorama dated 7 September 1936, National Archives and Records Administration, Seattle, Washington, USA. Bottom photo in each pair taken from 9,000 feet on 18 September 2018. Copyright 2018 John F. Marshall.

opportunities for abundant tree recruitment, particularly on more productive sites (Merschel et al. 2014, Johnston 2017) and during wet periods (Taylor 2000, Brown and Wu 2005, Brown 2006, Battaglia et al. 2018).

Simulations of wildfire and vegetation dynamics show that when fire is excluded from frequent-fire ecosystems, tree density increases; the proportion of fire-intolerant species increases; surface, ladder, and canopy fuels

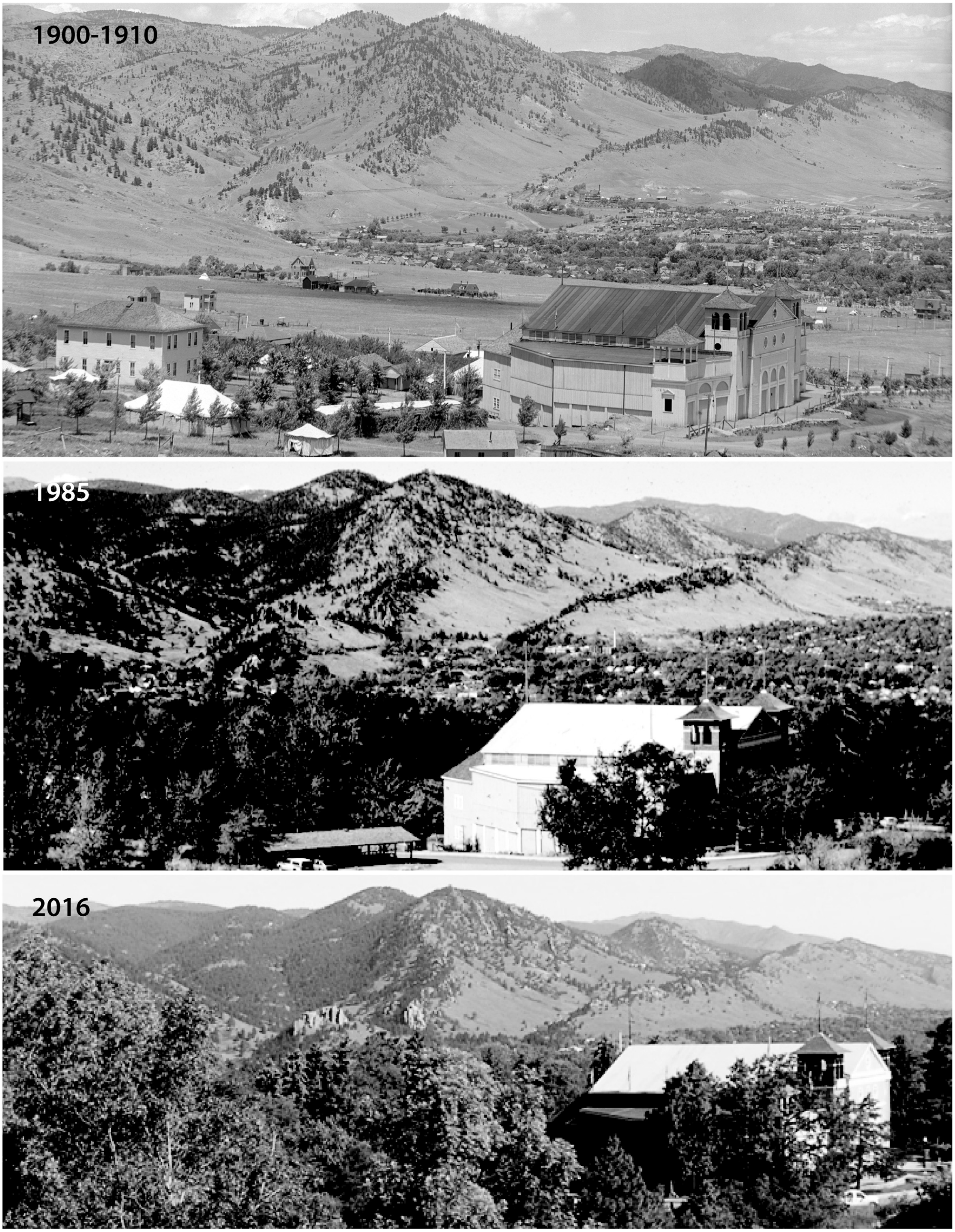


Fig. 6. Repeat photography from 1900 to 1910, 1985, and 2016 illustrates densification and expansion of ponderosa pine cover under fire exclusion in hills west of Boulder, Colorado (Veblen and Lorenz 1991). Photo credits: 1900–1910, Louis C. McClure Courtesy Denver Public Library, Western History Collection, MCC-306; 1985, T. T. Veblen and D. C. Lorenz; 2016, T. T. Veblen.

accumulate; and water available for forest growth declines (Wallin et al. 1996, Wimberly and Kennedy 2008, Diggins et al. 2010). These conditions can foster large and intense fires with effects that are not often observed in simulated historical ranges (Hann et al. 1997, Keane et al. 2009, 2018, Holsinger et al. 2014, Loehman et al. 2017, Haugo et al. 2019, Stockdale et al. 2019*b*). Correspondingly, forest succession and disturbance modeling projects lighter fuel loads and fewer high-intensity fires when departures from active fire regimes are low (King et al. 2008, Riggs et al. 2015).

Stands and landscapes with relatively intact or restored fire regimes (i.e., active fire regimes) provide insight into how historical forests and landscapes operate under contemporary climate and disturbance regimes (Cortés Montaño et al. 2012, Yocom Kent et al. 2017, Arizpe et al. 2021, Dewar et al. 2021, Murphy et al. 2021; and sources in Prichard et al. 2021: Table 2). Contemporary forests with relatively intact fire regimes experienced the climate variations of the 19th and 20th centuries, but do not exhibit changes in structure and composition comparable to fire-excluded forests (Stephens and Fulé 2005, Lydersen and North 2012, Pawlikowski et al. 2019). Similarly, forests with relatively intact fire regimes have not experienced the increased severity of disturbance events observed on comparable areas affected by fire exclusion (Rivera-Huerta et al. 2016, Murphy et al. 2021).

Broader impacts of fire regime departures

Modern wildfire suppression extinguishes essentially all fire starts except those that overwhelm fire suppression capacity and can only be extinguished when aided by a significant change in the weather (North et al. 2015, Moreira et al. 2020). Despite increasing suppression efforts, both area burned and area severely burned have increased as temperatures and widespread drought accelerated near the end of the 20th century (Westerling et al. 2006, Abatzoglou and Williams 2016, Parks and Abatzoglou 2020). Nonetheless, burned area in most forested ecosystems is still much lower than would be expected based on fire–climate relationships (Stephens et al. 2007, Fulé et al. 2012*b*, Marlon et al. 2012, Mallek et al. 2013, Parks et al. 2015, Taylor et al. 2016).

Contemporary fires burn in landscapes with greater forest density and connectivity, surface fuel accumulation, and proportion of small trees relative to larger, more fire-resistant trees, all of which contribute to more severe fires (Graham et al. 1999, 2004, Jain and Graham 2007). An eight-fold increase in annual area burned at high-severity occurred between 1985 and 2017 in western U.S. forests (Parks and Abatzoglou 2020), and fuels (i.e., live and dead vegetation) have been implicated as the primary driver of stand-replacing fire in most regions of the western United States (Steel et al. 2015, Parks et al. 2018). Departures from forest structures and compositions maintained by active fire regimes also contributed to uncharacteristically high levels and patterns of mortality during recent severe droughts (Bentz et al. 2010, Fettig et al. 2013, 2019, Stephens et al. 2018*a*). During those same droughts, however, stands with lower live basal area or density experienced lower tree mortality rates than stands with higher basal area or density for a given moisture regime, especially on drier sites (Rivera-Huerta et al. 2016, Young et al. 2017, Restaino et al. 2019).

High-severity fire is an essential component of many forested landscapes, not only through the provision of unique snag and complex early seral habitats (Swanson et al. 2011), but also through its influence on numerous other ecosystem functions, including nutrient and hydrological cycles and the rate and abundance of debris flow and sediment deposition (Bisson et al. 2003). However, constraints imposed on the relative abundance and patch sizes of high-severity fire by active fire regimes in dry, moist, and cold forests are also critical to maintaining the diverse and unique ecosystem characteristics of seasonally dry forested landscapes (Fig. 4, Table 1). Given widespread reductions in nonforest in the 20th century, some conversion of forest to nonforest area may aid recovery of ecosystem functions associated with active fire regimes, as seen where wildland fire was restored after nearly a century of fire exclusion (Boisramé et al. 2017*a*). Additionally, some cover type conversions are inevitable as landscapes adjust to a warming climate, perhaps particularly in southwestern North America (Falk 2013, Loehman et al. 2018, Field et al. 2020).

Studies of contemporary fires demonstrate, however, that high-severity fire is overrepresented in forests historically characterized by frequent low- to moderate-severity fire regimes (Table 2). Increased frequency of high-severity fire in these forest types is a concern for many reasons, including the likelihood that areas burned at high severity often reburn at high severity (Thompson et al. 2007, Lydersen et al. 2017, Prichard et al. 2017, Collins et al. 2018, Coop et al. 2020, Povak et al. 2020) even after a century of fire exclusion and forest succession (Taylor et al. 2020). Spatial patterns of high-severity fire in these forests are also a key departure of contemporary fire regimes (Hessburg et al. 1999*a*, 2015, 2019, Fulé et al. 2014, Reilly et al. 2017, Stevens et al. 2017). Even in forest types historically dominated by infrequent high-severity fire, fire severity patterns have likely changed given the suppression of most fire starts and absence of fires spreading in from adjacent forest and nonforest (Fig. 4; Perry et al. 2011, O'Connor et al. 2014, Johnson and Margolis 2019).

In landscapes historically dominated by frequent low- and moderate-severity fires, increases in high-severity fire are further reducing the abundance of large and old fire- and drought-tolerant trees (Table 2). These once prevalent trees, are currently rare and "endangered" (Stephens et al. 2016, Miller and Safford 2017, Reilly et al. 2018). Large and old fire- and drought-tolerant trees were heavily logged in the 20th century (Hessburg and Agee 2003, Brown and Cook 2006, Naficy et al.

TABLE 2. High-severity fire effects in recent fires exceed the pre-fire exclusion range of variation in landscapes historically dominated by frequent low- and moderate-severity fires.

Citation	Key findings	Forest type	Methods	Study area
Mallek et al. (2013)	In lower and middle elevation forests, area burned at low- to moderate-severity fire is substantially lower than expected while severity in recent fires is much higher than estimated for conditions prior to fire exclusion. Fires of all severities are at a deficit in upper elevation forests.	Lower (oak woodlands to ponderosa and Jeffrey pine), middle (mixed conifer), and upper (red fir and subalpine forest) elevation forests.	Compared fire severity distributions in modern (1984–2009) fires based on relative delta normalized burn ratio (RdNBR) with pre-fire exclusion fires based on average of LANDFIRE Biophysical Settings (BPS) and Stephens et al. (2007).	Sierra Nevada and southern Cascade Ranges, California
O'Connor et al. (2014)	Conversion of more than 80% of landscape from frequent low- to mixed-severity fire regime to one of infrequent moderate- to high-severity fire. Current high fuel loads shift climate drivers of fire behavior: (1) extreme drought no longer necessary for fire spread to mesic forest types and (2) antecedent moist conditions no longer necessary for spreading fires.	Pine and dry mixed conifer	Compared fire size and severity distributions in modern (1996 and 2004, RdNBR) fires with size and severity of fires prior to 1880 reconstructed from a gridded tree-ring sampling network.	Pinaleño Mountains, southeastern Arizona
Harris and Taylor (2015)	Increases in tree density, basal area, and fuels due to fire exclusion since 1899 shifted fire regime from frequent low severity to mixed severity.	Mixed conifer	Compared fire severity in 2013 (RdNBR) with fire severity prior to 1899 reconstructed from documentary records, radial growth of tree rings, fire-scars, and tree-age structure.	2013 Rim Fire, Yosemite National Park, California
Yocom-Kent et al. (2015)	Largest (>1,000 ha) high-severity patches in modern (2000–2012) fires exceeded those reconstructed for 1,400 ha study area; however, cannot rule out stand-replacing fire prior to mid-1700s	Mixed conifer and aspen	Compared high-severity fire patch size in modern (2000–2012) fires reconstructed from ground-truthing of satellite imagery with historical fires reconstructed from fire-scar and tree-age data.	North Rim, Grand Canyon National Park, Arizona
Fornwalt et al. (2016)	Tree(s) >200 yr old present in 4% area after fire compared to 70% before fire.	Unlogged ponderosa and ponderosa–Douglas-fir	Compared 2013 aerial imagery to pre-fire age structure in randomly selected polygons.	2002 Hayman fire, Colorado
Rivera-Huerta et al. (2016)	Following 30 yr of fire suppression, increasing high-severity patch size; fires remain easy to suppress and predominantly low.	Jeffrey pine and mixed conifer	Quantified area burned at high-severity in fires from the onset of fire suppression (roughly 1984) to 2010. RdNBR threshold of 652 indicates ≥90% reduction in basal area.	Baja California, Mexico
Bigio et al. (2010, 2017)	2002 Missionary Ridge fire was the most extensive and severe fire event in at least the past 2,600 yr in this steep, mountainous terrain.	Ponderosa and Gambel oak (*Quercus gambelii*) to mixed conifer	Compared fire-related deposition from debris flow and sediment-laden floods following the 2002 fire with alluvial-sediment records covering 3,000 yr.	2002 Missionary Ridge fire, San Juan Mountains, Colorado
Reilly et al. (2017)	High-severity fire effects in 23–26% of burned area in 1985–2010 exceeded expectations in most fire history studies.	Ponderosa pine and mixed conifer	Compared fire severity distributions for modern fires (1985–2010, RdNBR) with expected distributions derived from fire history studies; RdNBR burn severity thresholds were derived from pre- and post-fire CVS inventory data.	Oregon and Washington

Table 2. Continued

Citation	Key findings	Forest type	Methods	Study area
Safford and Stevens (2017) (Fig. 6 adapted from Miller and Safford 2008)	Area burned at high severity in modern fires exceeded estimates of area burned prior to European colonization.	Ponderosa and Jeffrey pine and mixed conifer	Compared modern fires (1984–2004, RdNBR) with Landfire BPS model estimates of high-severity fire extent prior to European colonization.	Sierra Nevada, California
Walker et al. (2018)	For areas that burned under extreme fire weather, sites lacking recent prior fire overwhelmingly converted to non-forest; more than half the total fire area is >50 m from surviving seed source.	Ponderosa and mixed conifer	Compared burn severity in 2011 (dNBR) on sites that had not burned in >100 yr with sites previously burned in prescribed fire and wildfire events that approximated fire frequency prior to fire exclusion.	2011 Las Conchas fire, northern New Mexico
Hagmann et al. (2019)	Stand-replacing fire effects in 23% of burned area in 1985–2015 compared to 6% in 1918.	Ponderosa pine, lodgepole pine, and mixed conifer	Compared extent of stand-replacing fire (RdNBR threshold of 962) for 1985–2015 fires (61,188 ha) with extent of burned area with no live trees >15 cm dbh following fires that burned >78,900 ha in 1918.	Pumice Plateau ecoregion, Oregon
Haugo et al. (2019)	High-severity fire effects in 36% of burned area in 1984–2015 exceeded 6–9% expected historically.	Frequent low-severity, FRG I	Compared area burned at high severity in modern (1984–2015, RdNBR) fires using previously validated thresholds for low, moderate, and high burn severity classes with simulated historical fire regime using BPS models in LANDFIRE.	Oregon and Washington
Nigro and Molinari (2019)	Average proportion burned at high severity in modern (2000–2016) fires more than 1.5 times greater than historical estimates; largest patch sizes larger than those recorded since 1900.	Ponderosa and Jeffrey pine and mixed conifer	Compared area burned at high severity in modern (2000–2016) fires using RdNBR threshold for ≥90% reduction in basal area with LANDFIRE BPS and relevant literature.	Sky island forests, southern California
Taylor et al. (2020)	In 2008, proportionally more mortality occurred in low and mid-elevation forests and less in high-elevation forests than in the 19th century.	Unlogged low and mid-elevation ponderosa pine, oak, and mixed conifer forests and high-elevation red fir forests.	Compare spatial patterns of fire severity in 2008 fire (RdNBR) burning under moderate weather with those of the late 19th century reconstructed from tree-ring and documentary records.	Cub Creek Research Natural Area, northern California

2010), and populations have continued to decline due to direct and indirect effects of drought stress, bark beetle outbreaks, and wildfire (Bentz et al. 2010, Fettig et al. 2013, 2019, McIntyre et al. 2015, Lydersen et al. 2017, Stephens et al. 2018*a*, Restaino et al. 2019, van Mantgem et al. 2020). Bark beetle outbreaks, accentuated by high forest density that exacerbates drought stress and facilitates infestation, continue to reduce average tree size and age in ponderosa and Jeffrey pine forests as bark beetles preferentially target larger individuals (Fettig et al. 2019, cf. Hood et al. 2020).

Recent trends in high-severity fire effects may contribute to further departures and present impediments to forest regeneration due to limitations on seed dispersal capacity and altered site conditions (Stevens-Rumann et al. 2018, Davis et al. 2019, Stevens-Rumann and Morgan 2019), particularly in the case of short interval reburns (Stephens et al. 2018*a*, Coop et al. 2020). Constraints on tree regeneration may be an inevitable consequence of a warming climate. Note, however, that regeneration in semiarid forest–steppe ecotones exhibited resilience to recent low-severity fires but not high-severity fires (Harris and Taylor 2020). Additionally, recent work shows that the biogeochemical impacts of high-severity fires are much longer-lasting than previously assumed, leading to concern that increased high-severity burning will negatively impact soil organic carbon and nutrient cycling (Dove et al. 2020).

Evaluating Evidence of Lack of Change

In this section, we review publications that suggest the preponderance of evidence misrepresents or overgeneralizes departures from active fire regimes. These publications then suggest that management actions aimed at recapturing the influence of abundant low- and moderate-severity fire lacks a sound ecological foundation. Over the past two decades, independent research groups have evaluated the methods and inferences proposed by these publications and documented multiple weaknesses. Despite demonstrated methodological biases and errors, new papers employing those methods, or results and conclusions derived from them, continue to pass peer review. To aid evaluation of this body of counter-evidence, we apply the same framework used above to evaluate evidence of change in forest conditions and fire regimes. We also synthesize peer-reviewed evaluations of the methods used in counter-evidence publications.

Misrepresented historical forest conditions

Publications based on novel methods for estimating historical forest density (Williams and Baker 2011) from early land surveys conducted by the General Land Office (GLO) have suggested that densities and fire severities of dry forests were higher and more variable than previously thought (Table 3). As described below, limitations of both GLO data and the methods used undermine this conclusion. Additionally, as Fulé et al. (2014) observed, existing research documented even greater heterogeneity in historical forest conditions, including higher densities, than was reconstructed from GLO data. Conflating high-frequency, low-severity fire regimes with homogeneity misrepresents the heterogeneity of those systems and disregards critical ecosystem functions associated with fine-scale spatial patterns in uneven-aged, predominantly open-canopy forests dominated by mature and old trees (Table 1).

Valid methods exist for deriving density estimates from spatial point patterns, such as GLO bearing trees (Cogbill et al. 2018). However, the extremely low sampling density of this national land survey limits reliable estimates to the average forest density for a large area. The typical spacing of 0.8 km between GLO survey points and a maximum of two or four trees per point yields a sample of, at most, eight trees per 260 ha. Levine et al. (2017) documented roughly 50% accuracy given a minimum of 50 GLO survey points (roughly 3,000 ha). Hanberry et al. (2011) documented accuracy of ±10% given GLO survey points in 10–20 townships (90,000–180,000 ha) depending on the number of bearing trees per point.

Thus, even when using independently validated methods, estimates of average density at such coarse spatial scales mask substantial heterogeneity in forest conditions at fine- and meso-scales. These records cover essentially all of the western United States, however, and can provide valuable insights into landscape change at coarse scales. For example, Knight et al. (2020) reconstructed average tree density for the floristically diverse Klamath Mountains at township (roughly 9,320 ha) resolution and documented substantial departures from historical conditions, including forest densification and loss of oak woodlands.

Williams and Baker (2011) proposed a method for estimating average tree density for three and six pooled GLO survey points (roughly 260 and 520 ha, respectively). However, due to the lack of a correction factor that accounts for the number of trees used to estimate density at individual sampling points, methods developed by Williams and Baker (2011) overestimated tree densities by 24–667% for contemporary stands with known densities (Levine et al. 2017, 2019). Levine et al. (2017, 2019) enabled independent evaluation of their methods and data by archiving all GLO estimator code and data on publicly accessible websites; data and code supporting Williams and Baker (2011) are not similarly accessible (Stephens et al. 2021). Independently validated methods for estimating tree density from point data were shown to yield estimates that were less biased (Levine et al. 2017) as well as more consistent with tree-ring reconstructions and less than half as large (Johnston et al. 2018) as those produced using Williams and Baker (2011) methods.

Density estimates based on Williams and Baker (2011) methods are also inconsistent with tree-ring reconstructions and early 20th-century timber inventory records for areas where the data overlap (Tables 3–5). Counter-evidence publications have suggested that tree-ring reconstructions might overrepresent the historical influence of low- to moderate-severity fire (Table 4) and that early timber inventories (which systematically sampled 10–20% of the area of relatively large landscapes) are biased, inaccurate records of historical tree densities (Table 3). Like all data sets, dendroecological reconstructions and early timber inventories have limitations. However, as described below and in Tables 3–5, independent research groups have tested methodological concerns about underrepresentation of high-severity fire effects and the capacity of early timber inventories to represent early 20th century forest conditions and shown them to be unfounded. Dendrochronological reconstructions and early timber inventories demonstrate consistency with each other and with other independent data sources (Scholl and Taylor 2010, Stephens et al. 2015, Hagmann et al. 2017, 2019).

When comparing study results, accounting for ecologically relevant differences in site conditions or methodologies is essential. However, counter-evidence publications consistently do not account for these differences (Tables 3–6). One of many such comparisons involves a study of ponderosa pine on two 1-ha plots each of which was intentionally selected because it contained >75 trees per hectare >250 yr old (Morrow 1985). Baker and Hanson (2017) compared average tree density in these two selectively sampled hectares (Morrow 1985) with average

Table 3. Publications presenting (1) counter-evidence asserting that forests were denser than previously thought and (2) evaluations of methods and inferences in counter-evidence publications.

Counter-evidence		Evaluation of counter-evidence	
Citations	Counter-premise	Citations	Implications of evaluation
Williams and Baker (2011) Baker and Williams (2018)	Novel methods provide estimates of tree density from point data, *i.e.*, General Land Office (GLO) records of bearing trees.	Levine et al. (2017, 2019)	Multiple existing plotless density estimators (PDE) provided less biased estimates than the PDE developed by Williams and Baker (2011), which overestimated known tree densities by 24–667% in contemporary stands.
		Knight et al. (2020)	Methods supported by PDE sampling theory and multiple accuracy assessments further demonstrate the potential for misrepresentation of historical tree density by biased estimators used at resolutions substantially smaller than the minimum recommended for ˜50% accuracy.
Williams and Baker (2012)	Historical forests were denser than previously documented.	Johnston et al. (2018)	Existing methods for estimating tree density from point data (Morisita 1957, Warde and Petranka 1981) yielded densities more consistent with tree-ring reconstructions and less than half as large as estimates using Williams and Baker (2011) methods.
Williams and Baker (2012) Baker (2015*a*, *b*, 2012, 2014)	Historical forests were denser than previously documented.	Hagmann et al. (2013, 2014, 2017, 2019), Collins et al. (2015), Stephens et al. (2015, 2018*c*), Battaglia et al. (2018), Johnston et al. (2018)	Consistent with the finding that Williams and Baker (2011) methods overestimate tree density (Levine et al. 2017, 2019, Johnston et al. 2018, Knight et al. 2020), early timber inventory records and tree-ring reconstructions for the same study areas documented substantially lower tree densities than those estimated using Williams and Baker (2011) methods.
Hanson and Odion (2016)	Managing for dense, old forest and high-severity fire is consistent with historical conditions.	Collins et al. (2016)	Fundamental errors compromise assertions about historical conditions, including: (1) inappropriate use of coarse-scale habitat maps and (2) inaccurate assumption that areas lacking timber volume in early inventories indicate past high-severity fire.
Odion et al. (2014), Baker (2015*a*, *b*) Baker and Hanson (2017)	Spatially extensive early timber inventories and bias in their use and interpretation misrepresent historical conditions.	Stephens et al. (2015), Collins et al. (2016), Hagmann et al. (2017, 2018, 2019)	Fundamental errors compromise conclusions, including: (1) use of previously discredited methods (Williams and Baker 2011) to estimate tree density from GLO data as a baseline comparison; (2) incorrect assumptions about the methodological accuracy of early timber inventories; (3) inappropriate comparisons of studies of vastly different spatial scales, forest types, and diameter limits; (4) unsubstantiated assessment of bias in the locations of early timber inventories; and (5) unwarranted assumptions about vegetation patterns as indicators of fire severity.

tree density for >50,000 ha of mixed-conifer forest from a systematic sample of 20% of the area in an early timber inventory (Hagmann et al. 2014). Williams and Baker (2011) and Baker and Williams (2018) also compared average tree density in these two selectively sampled hectares (Morrow 1985) with average density estimated for 520 ha from GLO land survey data. Average tree density in two selectively sampled 1-ha plots cannot credibly be assumed to represent average densities for the substantially larger areas in these comparisons, particularly given abundant documentation of fine- and meso-scale variation in historical forest and landscape structure (Table 1).

The low sampling density and, hence, low spatial resolution of GLO land survey data precludes analysis of the spatial patterns that influence disturbance severity or response to disturbance in forests and forested landscapes. Validated methods for deriving estimates of average tree density from GLO data may support conclusions about changes in average tree density or composition at broad spatial scales (e.g., Knight et al. 2020). However, objective conclusions about lack of change in forest conditions and fire regimes require additional lines of evidence. Multi-scale analysis of forest conditions and fire regimes (including spatial patterns in tree clumps, canopy gaps, forest successional types, physiognomic types, and stand-replacing fire) is essential to avoid misleading interpretations of the degree of ecosystem departures.

Misrepresented fire regimes

Counter-evidence publications have also posited that the high-severity component of contemporary wildfires is consistent with historical fire regimes based on the suggestions that high-severity fire was common historically (Tables 4 and 5) and modern wildfire severity is overestimated (Table 6). These assertions are

Table 4. Publications presenting (1) counter-evidence asserting that tree-ring reconstructions overestimate fire frequency and rotation and (2) evaluations of methods and inferences in counter-evidence publications.

Counter-evidence		Evaluation of counter-evidence	
Citations	Counter-premise	Citations	Implications of evaluation
Baker and Ehle (2001, 2003) Ehle and Baker (2003), Kou and Baker (2006*a*, *b*), Baker (2006, 2017), Dugan and Baker (2014)	Tree-ring reconstructions misrepresent historical fire regimes by overestimating fire frequency and extent because (1) unrecorded fires (e.g., fires that did not scar trees) increase uncertainty of mean fire interval (MFI); (2) interval between pith (origin) and first fire scar should be considered a fire-free interval and included in calculations of MFI; (3) targeted sampling of high scar densities biases MFI; (4) mean point fire interval (mean of intervals between fire scars weighted by the number of fire scars) may more accurately represent historical fire rotation than MFI (mean interval between all fire scars).	Collins and Stephens (2007)	Unrecorded fires (fire did not scar the tree) may contribute to underestimation, not overestimation, of fire frequency and extent in frequent fire systems. Probability of scarring decreased when intervals between successive fires were short in areas burned by up to four late 20th-century fires. Absence of scar does not indicate absence of fire.
		Brown and Wu (2005), Van Horne and Fulé (2006) Brown et al. (2008) Stephens et al. (2010), Yocom Kent and Fulé (2015) Meunier et al. (2019)	Including origin-to-first-scar interval erroneously inflates MFI. Not all trees that survive fire are scarred. As an ambiguous indicator of fire-free interval, it should not be included in calculations of MFI. Additionally, tree establishment may not indicate a stand-replacing disturbance in dry forests where regeneration is strongly associated with climate.
		Fulé et al. (2003) Van Horne and Fulé (2006) Farris et al. (2010, 2013) O'Connor et al. (2014)	Complete, systematic (gridded), and random sampling at stand, watershed, and mountain range scales have repeatedly demonstrated fire frequencies similar to those derived from targeted sampling within forest types and scales. In direct comparison studies, no evidence was found that targeted sampling of fire-scarred trees biased MFI estimates. Targeted sampling reconstructed fire parameters comparable to those derived from systematic sampling of both a subset of the trees and all trees in a study area and from independent 20th-century fire atlases.
		Farris et al.(2010) Huffman et al. (2015)	Rather than overestimating fire frequency as suggested in counter-premise papers, MFI may underestimate fire frequency, especially where small fires were abundant.
		Van Horne and Fulé (2006) Farris et al. (2013)	Composite mean fire intervals (CMFI, e.g., fires recorded on 25% of samples) are relatively stable across changes in sample area or size. See the section on "Underestimated historical fire frequency" for a more detailed summary of CMFI and the highly problematic and inherently biased alternatives proposed in counter-evidence publications.

compromised by methodological errors leading to underestimation of historical fire frequency, overestimation of historical fire severity, and underestimation of contemporary fire severity, as described in this section. Additionally, without consideration of all dimensions of a fire regime, one cannot objectively conclude that ecologically relevant departures have not occurred. For example, while fire regimes may not differ in one dimension (e.g., abundance of high-severity fire), they may differ in other dimensions (e.g., size or complexity of patches of high-severity fire). Similarly, while the dominant fire severity class (e.g., moderate) may not have changed for a given area, median percent mortality may have (e.g., a shift from 30% to 70%).

Underestimated historical fire frequency.—Dendroecological methods for reconstructing spatial point patterns of fire history have well-documented strengths and limitations (Stokes and Dieterich 1980, Baisan and Swetnam 1990, Falk et al. 2011, Daniels et al. 2017). Fire scars record low-severity (non-lethal) fire at a specific place and time; however, absence of a scar on nearby trees may indicate either that the area did not burn or that it burned without scarring (i.e., absence of evidence is not evidence of absence; Fig. 7c). A common approach to overcoming this uncertainty is to composite fire scar dates from multiple trees. As more trees are sampled, the probability of detecting additional fires increases and eventually plateaus (Falk and Swetnam 2003). As trees are sampled across larger landscapes, the composite mean fire interval (CMFI) may be reduced as more fire-scar dates are found, especially in forests that historically experienced numerous relatively small fires (Collins and Stephens 2007). To avoid overestimating fire return intervals based on point sampling, researchers recommend (1) limiting compositing of fire dates to relatively small areas where fuel and topographic conditions would likely have resulted in generally uniform burning conditions; (2) collecting numerous samples to saturate the list of fire dates, reporting the point fire interval, and demonstrating a sampling plateau (Falk and Swetnam 2003, Van Horne and Fulé 2006); (3) using minimum

Table 5. Publications presenting (1) counter-evidence asserting that modern wildfires are not unlike historical fires because severity of historical fires is underestimated and (2) evaluations of methods and inferences in counter-evidence publications.

Counter-evidence		Evaluation of counter-evidence	
Citations	Counter-premise	Citations	Implications of evaluation
Shinneman and Baker (1997)	Based on early forest inventory age data sets, "nonequilibrium" areas of extensive, high-severity fires in the Black Hills led to landscapes dominated by dense, closed-canopy forests.	Brown (2006)	Tree-ring reconstructions of ponderosa pine forest age structures and fire regimes across the Black Hills found synchronous regional tree recruitment largely in response to pluvials and longer intervals between surface fires, especially during the late 1700s/early 1800s, which is when early inventory data report similar patterns of recruitment. No evidence of crown fires was found in relation to past fire dates.
Baker et al. (2007)	Most ponderosa pine forests in the Rocky Mountains were capable of supporting high-severity crown fires as well as low-severity surface fires.	Brown et al. (2008)	Tree-ring reconstruction of ponderosa pine forests in the Black Hills of South Dakota (included in Baker et al. 2007) demonstrated that roughly 3.3% of the study area burned as crown fire between 1529 and 1893; however, tree density in most stands in 1870 could not have supported crown fire.
Williams and Baker (2012), Baker (2012, 2014)	Fire severity inferred from tree density by size class estimated from GLO bearing trees (Williams and Baker 2011) and surveyors' descriptions suggests low-severity fire dominated only a minority of ponderosa and mixed-conifer forests.	Levine et al. (2017, 2019)	Plotless density estimator used by Williams and Baker (2011) overestimated known tree densities due to a scaling factor that does not correct for the number of trees sampled and therefore systematically underestimates the area per tree relationship.
		Fulé et al. (2014), Merschel et al. (2014), O'Connor et al. (2017)	Substantial errors of method and interpretation invalidate inferences about historical fire severity. These include (1) tree size is an ambiguous indicator of tree age; (2) tree regeneration is an ambiguous indicator of disturbance severity, particularly in dry forests where climate conditions strongly influence regeneration; and (3) lack of direct documentary evidence (e.g., primary observation) of extensive crown fire in historical ponderosa pine forests has been widely noted for nearly 90 yr.
		Stephens et al. (2015), Huffman et al. (2015), Miller and Safford (2017), Hagmann et al. (2019)	Multi-proxy records documented substantially lower levels of high-severity fire in ponderosa and Jeffrey pine and mixed-conifer forests in overlapping study areas.
Baker (2012), Baker and Hanson (2017)	Estimates of area burned at high severity in Hessburg et al. (2007) validate estimates derived using Williams and Baker (2011) methods.	Hagmann et al. (2018), Spies et al. (2018*a*)	Inappropriate comparisons are not validation. Baker (2012) limited assessment of high-severity fire to tree mortality in dry forests whereas Hessburg et al. (2007) estimated high-severity fire in the dominant cover type whether that be grass or tree for "moist and cold forest" type, with lesser amounts of dry forests
Odion et al. (2014)	Modern, high-severity crown-fires are within historical range of variation. Inferred fire severity from current tree-age data for unmanaged forests in the U.S. Forest Service Inventory and Analysis (FIA) program. Compared inferences about modern fire severity to estimates of historical forest conditions and fire severity inferred using Williams and Baker (2011) methods.	Fulé et al. (2014), Levine et al. (2017, 2019), Knight et al. (2020)	Overestimation of historical tree density and unsupported inferences of fire severity from GLO records weaken conclusions based on Williams and Baker (2011) methods.
		Stevens et al. (2016)	Substantial errors of method and interpretation invalidate inferences about historical fire severity. These include (1) FIA stand age variable does not reflect the large range of individual tree ages in the FIA plots and (2) recruitment events are not necessarily related to high-severity fire occurrence.
		Spies et al. (2018*a*, *b*)	In contradiction of the counter-premise, Odion et al. documented only three patches of high-severity fire larger than >1,000 ha in Oregon and Washington in the early 1900s, which account for 1% of the area of historical low-severity fire regime managed under the Northwest Forest Plan.

TABLE 5. Continued

Counter-evidence		Evaluation of counter-evidence	
Citations	Counter-premise	Citations	Implications of evaluation
Baker and Hanson (2017)	Stephens et al. (2015) underrepresented the historical extent of high-severity fire in their interpretation of surveyor notes in early timber inventory.	Hagmann et al. (2018)	Substantial errors of method and interpretation invalidate inferences about the historical extent of high-severity fire. Inferences were based on (1) inappropriate assumptions about the size and abundance of small trees given the ambiguity of data describing small trees in the 1911 inventory, (2) averaging of values derived from different areas and vegetation classifications, and (3) inappropriate assumption that the presence of chaparral (common on sites with thin soils and high solar radiation) indicates high-severity fire.

sample depths to account for periods when fire records may be missing; and (4) proportional filtering of fire dates to distinguish smaller from larger fires (Swetnam and Baisan 1996).

The efficacy of these methods has been repeatedly demonstrated, often through direct testing of criticisms raised in counter-evidence publications (Table 4). For example, to evaluate Baker and Ehle (2001) assertions that targeted sampling of high fire scar densities biases MFI, Van Horne and Fulé (2006) compared targeted, random, and grid-based sampling of fire-scarred trees to a census of all fire-scarred trees (n = 1,479) in a 1-km^2 area. Given a minimum sample size of 50 trees (3%), all methods accurately reproduced the mean fire interval of the census of all fire-scarred trees. Farris et al. (2013) also found a high degree of accuracy across multiple sampling regimes. Similarly, Farris et al. (2010) tested Baker and Ehle (2001) assertions that fire-scar histories overestimated fire frequency by giving undue importance to small fires. First, Farris et al. (2010) demonstrated that spatially distributed fire-scar samples accurately reconstructed the existing spatial and temporal record of mapped fire events >100 ha that occurred from 1937 to 2000. Contrary to Baker and Ehle (2001) assertions, Farris et al. (2010) found that fires <100 ha were more common in the record of mapped fire events than suggested by dendrochronological reconstruction. Fire-scar records for hundreds of studies across western North America are archived on publicly accessible databases (Falk et al. 2011), which enables independent evaluation of methods and inferences. Fire-scar data supporting counter-evidence publications (Tables 4 and 5) are not similarly accessible.

Compositing of fire-scar records has proven to be a reliable, repeatable, and robust method (Heyerdahl et al. 2001, Fulé et al. 2003, Taylor and Skinner 2003, Van Horne and Fulé 2006, Hessl et al. 2007, Farris et al. 2010, 2013, O'Connor et al. 2014*a*). However, counter-evidence publications present and support the use of alternative methods that are problematic to calculate and biased (Table 4). For example, Kou and Baker (2006*a*) proposed an "all-tree fire interval" (ATFI) metric that includes a "scarring fraction" (SF, estimated fraction of unscarred trees) to derive a "population mean fire interval" (PMFI). Few studies have tried to estimate SF (Kou and Baker 2006*a*); thus, few estimates of ATFI are available (Baker 2017). Additionally, as acknowledged by Kou and Baker (2006*a*: Accessory Publication), ATFI will always be much longer than any MFI, even for non-composited MFIs based on individual trees.

ATFI and SF are inaccurate indicators of historical fire occurrence. ATFI and SF depend on the false assumption that absence of scarring indicates absence of fire (Table 4). Reconstructing SF for each fire in a historical record is not feasible given that scarring can vary considerably with variations in weather and live and dead fuels between and within individual fires (Fig. 7a). Studies that have estimated SF (cited in Baker 2017) used data from recent fires that burned after a century or more of fire exclusion (Fig. 1) and are, therefore, not representative of historical fuel or fire behavior conditions. Additionally, ATFI inflates mean fire intervals by equating tree age with the period of fire regime analysis, thereby including origin-to-first-scar and time-since-last-fire intervals (Table 4, Fig. 7b). Abundant evidence indicates that origin-to-first-scar intervals are not reliable indicators of fire-free intervals and should be omitted from calculations of MFI (Table 4). Similarly, time-since-last-fire intervals that overlap more than a century of fire exclusion (Fig. 1) are not credible representations of fire frequency in active fire regimes (Figs. 1, 7).

Overestimated historical fire severity.—The indicators of high-severity fire events used in counter-evidence publications (e.g., average stand age, abundance of small trees, and presence of shrub fields) are ambiguous given ample viable alternative explanations for those conditions, as described below and in Table 5. High tree regeneration densities do not necessarily indicate prior fire events, a concept well-documented by the densification that has occurred in the absence of fire during the long 19th- to 21st-century period of fire exclusion (Hessburg and Agee 2003, Fulé et al. 2014, Merschel et al. 2014, O'Connor et al. 2017). Nonetheless, publications from Shinneman and Baker (1997) to those based on Williams and Baker (2011) use this metric to infer high-severity fire extent (Table 5). Climatic drivers of regeneration are often dis-associated with disturbance events (Brown and Wu

TABLE 6. Publications presenting (1) counter-evidence asserting that modern wildfires are comparable to historical fires because severity of modern fires is overestimated and (2) evaluations of methods and inferences in counter-evidence publications.

Counter-evidence		Evaluation of counter-evidence	
Citations	Counter-premise	Citations	Implications of evaluation
Odion and Hanson (2006)	High-severity fire was rare in recent fires in the Sierra Nevada based on analysis of Burned Area Emergency Response (BAER) soil burn severity maps.	Safford et al. (2008)	BAER maps greatly underestimate stand-replacing fire area and heterogeneity in burn severity for vegetation. BAER maps are soil burn-severity maps, not vegetation burn-severity maps.
Hanson et al. (2009)	Changes in conservation strategies for Northern Spotted Owl (NSO) were unwarranted due to overestimation of high-severity fire in the NSO recovery plan.	Spies et al. (2010)	Use of a higher relative delta normalized burn ratio (RdNBR) threshold substantially increased misclassification errors and reduced estimates of high-severity fire extent. Hanson et al. (2009) used an RdNBR threshold of 798 rather than 574 as recommended in the literature (Miller et al. 2009) they cited as the source of the threshold used.
Williams and Baker (2012)	Severity distributions in recent fires do not depart from historical.	Steel et al. (2015), Guiterman et al. (2015), Reilly et al. (2017), Steel et al. (2018)	Extent and spatial patterns of fire severity in some recent fires have departed from pre-fire exclusion range of variation for some forest types.
Hanson and Odion (2014)	Previous assessments overestimate extent of high-severity fire in modern fires.	Safford et al. (2015)	Use of coarse-scale, highly inaccurate, and geographically misregistered vegetation map and averaging across unrelated vegetation types and diverse ownerships undermine confidence in Hanson and Odion (2014).

2005, North et al. 2005*a*, Brown 2006, Brown et al. 2008, Swetnam and Brown 2010, Heyerdahl et al. 2014). Moreover, widespread livestock grazing promoted abundant regeneration by exposing mineral soil, reducing competition from grasses and herbs for resources (Rummell 1951, McKelvey and Johnston 1992, Hessburg and Agee 2003), and reducing fire spread by disrupting fuel continuity (Savage and Swetnam 1990, Belsky 1992, Belsky and Blumenthal 1997, Swetnam et al. 2016). Similarly, while shrub fields may be a legacy of type conversion after high-severity forest fires, other factors also maintained shrubs and constrained forest development, including frequent fire (Knapp et al. 2013, Guiterman et al. 2018), Indigenous resource management (Marks-Block et al. 2021), and biophysical conditions, e.g., thin soils and high solar radiation (Stephens et al. 2015). Thus, multi-proxy evidence and meta-analyses are often needed to reconstruct site history and more credibly evaluate changes to forest conditions.

Other methodological errors also contribute to overestimation of historical fire severity in counter-evidence publications (Table 5). As described above, inferences based on historical tree densities estimated from GLO land survey data using Williams and Baker (2011) methods warrant reconsideration given methodological errors documented by multiple independent research groups (Table 3). Further, estimates of high-severity fire extent in Hessburg et al. (2007) do not validate estimates based on Williams and Baker (2011) methods because they are not comparable (Spies et al. 2018*b*: 132–137), despite assertions to the contrary (Table 5). Estimates of area burned at high-severity using Williams and Baker (2011) methods are derived from inferred tree mortality while estimates derived from early aerial imagery (Hessburg et al. 2007) reflect mortality of the dominant cover type whether it be tree, shrub, or grass. Similarly, the use of average stand-age in contemporary Forest Inventory and Analysis (FIA) data by Odion et al. (2014) compromises inferences about the historical extent of high-severity fire due to the failure of this metric to account for the presence of older trees in the plot and the fact that tree age is not a reliable indicator of high-severity fire (Stevens et al. 2016).

Underestimated contemporary fire severity.—Estimates of contemporary fire severity in counter-evidence publications are compromised by non-standard definitions of high-severity fire and the capacity of the data to produce credible estimates of high-severity fire (Table 6). Odion and Hanson (2006) used data sets not designed to measure tree mortality (e.g., Burned Area Emergency Response data sets, BAER), which precludes ecologically meaningful comparisons with studies that classify fire severity as percentage of tree basal area or canopy cover killed (Safford et al. 2008). Hanson et al. (2009) used a higher severity threshold than recommended in the literature cited in support of their methods (Miller et al. 2009*a*), which yielded lower estimates of area burned by high-severity fire and weakened inferences based on comparisons with studies using the regionally calibrated threshold (Spies et al. 2010). Hanson and Odion (2014) suggested that previous assessments had

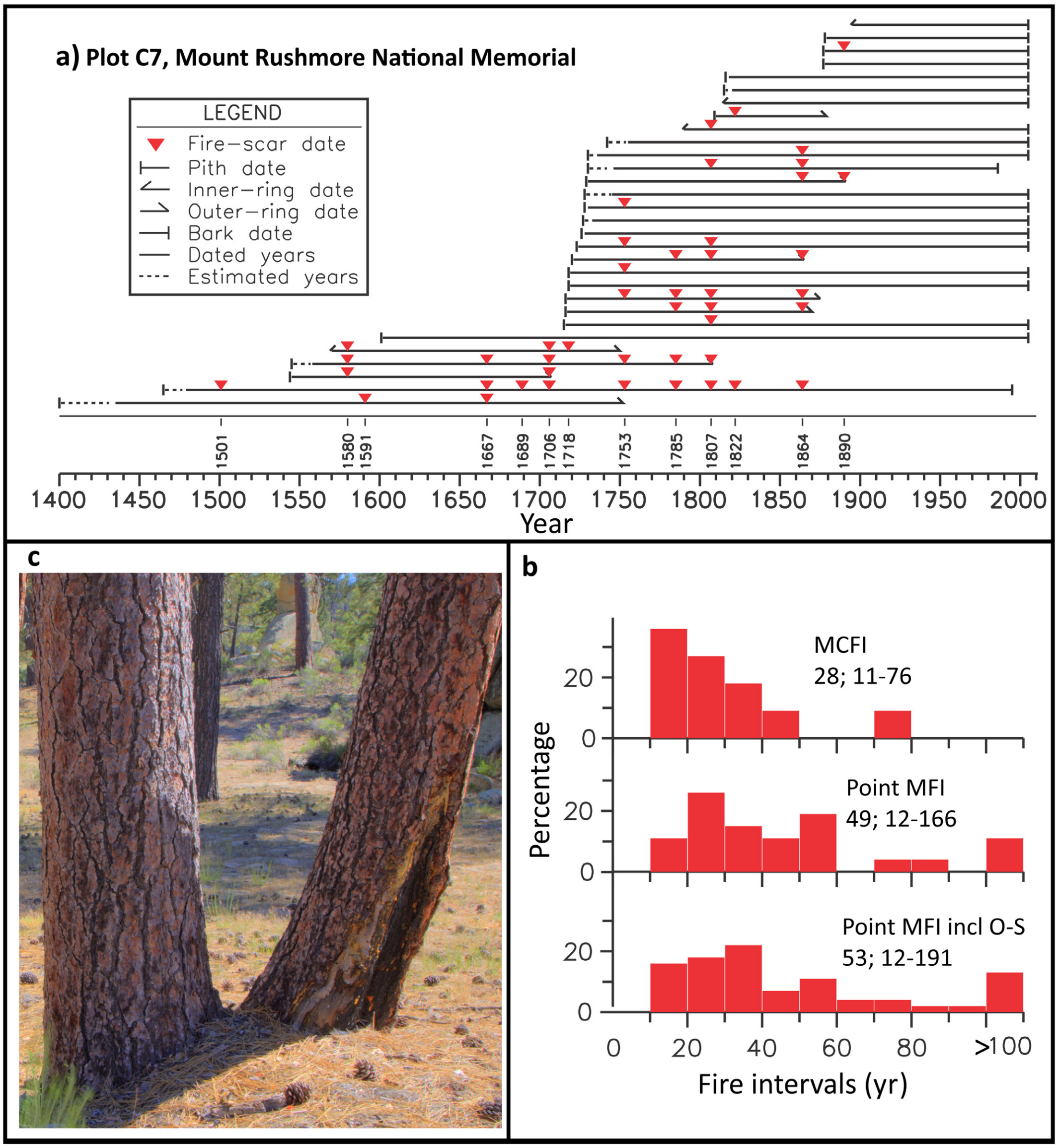


FIG. 7. A comparison of mean fire interval calculations using a fire history plot from Mount Rushmore National Memorial (Brown et al. 2008). (a) Fire-demography diagram of trees collected from *n*-tree variable radius plot. Data are cross-dated results from the 30 live (≥20 cm dbh) and dead trees nearest to a randomly selected grid point. Horizontal lines represent time spans of individual trees. Plot area is 0.11 ha, determined as a circular plot with radius of distance to farthest tree sampled. (b) Mean and range of fire intervals for this plot estimated by different methods. Top panel shows mean composite fire intervals (MCFI) using scar-to-scar intervals composited across all trees in the plot from 1580 to 1890 (11 total intervals). Fire dates used for interval calculation were those with minimum sample depth of five trees because of possible missing fire-scar records with fewer trees (i.e., the period between 1501 and 1580). Middle panel shows point mean fire intervals (point MFI) using scar-to-scar intervals recorded on all trees (27 total intervals). Bottom panel shows point MFI including origin-to-first scar (O-S) intervals on individual trees (45 total intervals). Including time-since-last fire intervals would further increase Point MFI. (c) An example of an unscarred tree of approximately the same age as a close neighbor with 14 fire scars. In a plot area of only 0.11 ha (panel a), all trees must have experienced fire at or very close to their stems for all fire dates listed but did record the event as a fire scar.

overestimated the extent of high-severity fire in modern fires; however, the use of a coarse-scale, highly inaccurate, and geographically misregistered vegetation map and averaging across unrelated vegetation types and diverse ownerships undermine confidence in this suggestion (Safford et al. 2015).

Misrepresented breadth and depth of change

A vast body of research progressively developed over more than a century, conducted at multiple spatial scales, and drawing on numerous intersecting lines of evidence underpins current scientific understanding of

the effects of prolonged fire exclusion on contemporary fire regimes and forest conditions. Given that evidence of change may not be apparent at all spatial scales or in all aspects of forest conditions and fire regimes, the conclusion that pattern-process interactions in fire-excluded forested landscapes have not departed from those characterizing active fire regimes requires strong evidence from multi-scale, multi-dimensional, multi-proxy evaluations. As demonstrated in the multiple independent assessments reviewed here (Tables 3–6), inferences supported by these counter-evidence publications are weakened by multiple methodological errors and warrant critical reevaluation. We conclude that these counter-evidence publications do not meet minimum standards for “best available science” to inform land and resource management on public lands (Esch et al. 2018).

Conclusions and Management Implications

Based on the strength of evidence, there can be little doubt that the long-term deficit of abundant low- to moderate-severity fire has contributed to modification of seasonally dry forested landscapes across western North America. The magnitude of change in fire regimes and the resultant increases in forest density and fuel connectivity have increased the vulnerability of many contemporary forests to seasonal and episodic increases in drought and fire, exacerbated by rapid climate warming. While some ecosystems within these landscapes have been less directly altered by fire exclusion, they may be indirectly affected by alteration of the surrounding landscape and consequent changes to ecosystem processes, including disturbance and hydrological regimes. These substantial departures as well as on-going wildfire exclusion threaten numerous social and ecological values, including quantity and quality of water supply, stability of carbon stores, and air quality (Stephens et al. 2020), as well as culturally important resources and food security (Norgaard 2014, Sowerwine et al. 2019).

Long-term fire exclusion leads to the loss of informational (species life history traits) and material (biotic and abiotic structures such as seeds and nutrients) legacies (Johnstone et al. 2016) that may compromise fire-dependent diversity and the capacity of forested ecosystems to resist or recover after wildfires, especially under climate change (Franklin et al. 2000, Reilly et al. 2019, Krawchuk et al. 2020). Among these legacies are mature and old trees, in particular, open-canopy forests of mature and old conifers and hardwoods, which provide unique ecosystem functions and which were once substantially more prevalent (Spies et al. 2006, Kolb et al. 2007, Long et al. 2015, Franklin et al. 2018, Long et al. 2018, Hanberry and Dumroese 2020). As climate continues to warm and burned area increases, early seral habitat will likely be created in abundance. However, recapturing the once extensive influence of the low- and moderate-severity fires that shaped and maintained these ecosystems for millennia requires a paradigm shift from strategies favoring fire suppression to those favoring fire-adapted forests and communities (reviewed by Hessburg et al. 2021, Prichard et al. 2021).

Perpetuating invalidated methods and inferences based on them fosters confusion and controversy, which undermine scientific credibility and impede the development of relevant and timely policy and management options. For example, counter-evidence reviewed here was used to support contentious conclusions in a meta-analysis of the impacts of high-severity fire on California Spotted Owls (*Strix occidentalis occidentalis*; Lee 2018). The authors of many of the studies included in that meta-analysis subsequently demonstrated methodological weaknesses in the meta-analysis that undermine those conclusions (Jones et al. 2020*a*). Unwarranted uncertainty about the use of high-severity burn areas by California Spotted Owls (Jones et al. 2019, Peery et al. 2019) has detrimentally impacted the management of this sensitive species (Stephens et al. 2019, Jones et al. 2020*b*). Objective scientific evaluation can aid in differentiating warranted from unwarranted uncertainties and enable timely paradigm shifts to policies and management actions that favor fire- and climate-adapted forests and human communities.

Acknowledgments

We are grateful for funding from California Department of Forestry and Fire Protection; Ecological Restoration Institute; Washington State Department of Natural Resources; The Wilderness Society; The Nature Conservancy of Oregon; Conservation Northwest; USDA-FS, Pacific Northwest and Pacific Southwest Research Stations; and USDI-FWS Oregon Fish and Wildlife Office. Constructive reviews from K. Norman Johnson, Cameron E. Naficy, James Rosen, Frank K. Lake, Collin Haffey, and two anonymous reviewers substantially improved our manuscript.

Disclaimer

The findings and conclusions in this article are those of the author(s) and do not necessarily represent the views of the U.S. Fish and Wildlife Service or the USDA Forest Service.

Literature Cited

Abatzoglou, J. T., and A. P. Williams. 2016. Impact of anthropogenic climate change on wildfire across western US forests. Proceedings of the National Academy of Sciences USA 113:11770–11775.

Addington, R. N., et al. 2018. Principles and practices for the restoration of ponderosa pine and dry mixed-conifer forests of the Colorado Front Range. RMRS-GTR-373. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Agee, J. K. 1996. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C., USA.

Agee, J. K. 1998. The landscape ecology of western forest fire regimes. Northwest Science 72:24–34.

Allen, C. D., R. S. Anderson, R. B. Jass, J. L. Toney, and C. H. Baisan. 2008. Paired charcoal and tree-ring records of

high-frequency Holocene fire from two New Mexico bog sites. International Journal of Wildland Fire 17:115–130.
Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, and J. T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications 12:1418–1433.
Amoroso, M. M., L. D. Daniels, M. Bataineh, and D. W. Andison. 2011. Evidence of mixed-severity fires in the foothills of the Rocky Mountains of west-central Alberta, Canada. Forest Ecology and Management 262:2240–2249.
Arizpe, A. H., D. A. Falk, C. A. Woodhouse, and T. W. Swetnam. 2021. Widespread fire years in the US–Mexico Sky Islands are contingent on both winter and monsoon precipitation. International Journal of Wildland Fire 29:1072–1087.
Baisan, C. H., and T. W. Swetnam. 1990. Fire history on a desert mountain range: Rincon Mountain Wilderness, Arizona, U.S.A. Canadian Journal of Forest Research 20:1559–1569.
Baker, W. L. 2006. Fire history in ponderosa pine landscapes of Grand Canyon National Park: is it reliable enough for management and restoration? International Journal of Wildland Fire 15:433–437.
Baker, W. L. 2012. Implications of spatially extensive historical data from surveys for restoring dry forests of Oregon's eastern Cascades. Ecosphere 3:1–39.
Baker, W. L. 2014. Historical forest structure and fire in Sierran mixed-conifer forests reconstructed from general land office survey data. Ecosphere 5:1–70.
Baker, W. L. 2015*a*. Historical northern spotted owl habitat and old-growth dry forests maintained by mixed-severity wildfires. Landscape Ecology 30:655–666.
Baker, W. L. 2015*b*. Are high-severity fires burning at much higher rates recently than historically in dry-forest landscapes of the Western USA? PLoS ONE 10:e0136147.
Baker, W. L. 2017. Restoring and managing low-severity fire in dry-forest landscapes of the western USA. PLoS ONE 12: e0172288.
Baker, W. L., and D. Ehle. 2001. Uncertainty in surface-fire history: the case of ponderosa pine forests in the western United States. Canadian Journal of Forest Research 31:1205–1226.
Baker, W. L., and D. Ehle. 2003. Uncertainty in fire history and restoration of ponderosa pine forests in the western United States. Pages 319–333 *in* P. N. Omi, and L. A. Joyce, editors. Fire, fuel treatments, and ecological restoration: Conference proceedings. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.
Baker, W. L., and C. T. Hanson. 2017. Improving the use of early timber inventories in reconstructing historical dry forests and fire in the western United States. Ecosphere 8:e01935.
Baker, W. L., T. T. Veblen, and R. L. Sherriff. 2007. Fire, fuels and restoration of ponderosa pine–Douglas fir forests in the Rocky Mountains, USA. Journal of Biogeography 34:251–269.
Baker, W. L., and M. A. Williams. 2018. Land surveys show regional variability of historical fire regimes and dry forest structure of the western United States. Ecological Applications 28:284–290.
Balch, J. K., B. A. Bradley, J. T. Abatzoglou, R. C. Nagy, E. J. Fusco, and A. L. Mahood. 2017. Human-started wildfires expand the fire niche across the United States. Proceedings of the National Academy of Sciences USA 114:2946–2951.
Barros, A. M. G., A. A. Ager, M. A. Day, M. A. Krawchuk, and T. A. Spies. 2018. Wildfires managed for restoration enhance ecological resilience. Ecosphere 9:e02161. https://doi.org/10.5751/ES-08917-220124.
Barros, A. M. G., et al. 2017. Spatiotemporal dynamics of simulated wildfire, forest management, and forest succession in central Oregon, USA. Ecology and Society 22:24.
Battaglia, M. A., B. Gannon, P. M. Brown, P. J. Fornwalt, A. S. Cheng, and L. S. Huckaby. 2018. Changes in forest structure since 1860 in ponderosa pine dominated forests in the Colorado and Wyoming Front Range, USA. Forest Ecology and Management 422:147–160.
Beaty, R. M., and A. H. Taylor. 2009. A 14,000 year sedimentary charcoal record of fire from the northern Sierra Nevada, Lake Tahoe Basin, California, USA. Holocene 19:347–358.
Belsky, A. J. 1992. Effects of grazing, competition, disturbance and fire on species composition and diversity in grassland communities. Journal of Vegetation Science 3:187–200.
Belsky, A. J., and D. M. Blumenthal. 1997. Effects of livestock grazing on stand dynamics and soils in upland forests of the interior west. Conservation Biology 11:315–327.
Bentz, B. J., J. Régnière, C. J. Fettig, E. M. Hansen, J. L. Hayes, J. A. Hicke, R. G. Kelsey, J. F. Negrón, and S. J. Seybold. 2010. Climate change and bark beetles of the western United States and Canada: direct and indirect effects. BioScience 60:602–613.
Betancourt, J. L., T. R. V. Devender, and P. S. Martin. 1990. Packrat middens: the last 40,000 years of biotic change. University of Arizona Press, Tucson, Arizona, USA.
Bigio, E., T. W. Swetnam, and C. H. Baisan. 2010. A comparison and integration of tree-ring and alluvial records of fire history at the Missionary Ridge Fire, Durango, Colorado, USA. Holocene 20:1047–1061.
Bigio, E. R., T. W. Swetnam, and P. A. Pearthree. 2017. Late Holocene fire–climate relationships of the western San Juan Mountains, Colorado. International Journal of Wildland Fire 26:944–962.
Bisson, P. A., B. E. Rieman, C. Luce, P. F. Hessburg, D. C. Lee, J. L. Kershner, G. H. Reeves, and R. E. Gresswell. 2003. Fire and aquatic ecosystems of the western USA: current knowledge and key questions. Forest Ecology and Management 178:213–229.
Bohlman, G. N., C. N. Skinner, and H. D. Safford. 2021. Natural range of variation (NRV) for yellow pine and mixed conifer forests in northwestern California and southwestern Oregon. General Technical Report PSW-GTR-273. USDA Forest Service, Pacific Southwest Research Station, Albany, California, USA.
Boisramé, G. F. S., S. E. Thompson, M. Kelly, J. Cavalli, K. M. Wilkin, and S. L. Stephens. 2017*a*. Vegetation change during 40 years of repeated managed wildfires in the Sierra Nevada, California. Forest Ecology and Management 402:241–252.
Boisramé, G., S. Thompson, B. Collins, and S. Stephens. 2017*b*. Managed wildfire effects on forest resilience and water in the Sierra Nevada. Ecosystems 20:717–732.
Boisramé, G. F. S., S. E. Thompson, C. Tague, and S. L. Stephens. 2019. Restoring a natural fire regime alters the water balance of a Sierra Nevada catchment. Water Resources Research 55:5751–5769.
Bowman, D. M. J. S., et al. 2009. Fire in the earth system. Science 324:481–484.
Bradstock, R. A., M. Bedward, B. J. Kenny, and J. Scott. 1998. Spatially-explicit simulation of the effect of prescribed burning on fire regimes and plant extinctions in shrublands typical of south-eastern Australia. Biological Conservation 86:83–95.
Brown, P. M. 2006. Climate effects on fire regimes and tree recruitment in Black Hills ponderosa pine forests. Ecology 87:2500–2510.
Brown, P. M., and B. Cook. 2006. Early settlement forest structure in Black Hills ponderosa pine forests. Forest Ecology and Management 223:284–290.

Brown, P. M., C. Gentry, and Q. Yao. 2019. Historical and current fire regimes in ponderosa pine forests at Zion National Park, Utah: restoration of pattern and process after a century of fire exclusion. Forest Ecology and Management 445:1–12.

Brown, P. M., C. L. Wienk, and A. J. Symstad. 2008. Fire and forest history at Mount Rushmore. Ecological Applications 18:1984–1999.

Brown, P. M., and R. Wu. 2005. Climate and disturbance forcing of episodic tree recruitment in a southwestern ponderosa pine landscape. Ecology 86:3030–3038.

Bryant, T., K. Waring, A. Sánchez Meador, and J. B. Bradford. 2019. A framework for quantifying resilience to forest disturbance. Frontiers in Forests and Global Change 2:56.

Chavardès, R. D., and L. D. Daniels. 2016. Altered mixed-severity fire regime has homogenised montane forests of Jasper National Park. International Journal of Wildland Fire 25:433–444.

Chavardès, R. D., L. D. Daniels, Z. Gedalof, and D. W. Andison. 2018. Human influences superseded climate to disrupt the 20th century fire regime in Jasper National Park, Canada. Dendrochronologia 48:10–19.

Churchill, D. J., G. C. Carnwath, A. J. Larson, and S. A. Jeronimo. 2017. Historical forest structure, composition, and spatial pattern in dry conifer forests of the western Blue Mountains, Oregon. PNW-GTR-956. Page 93. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Churchill, D. J., A. J. Larson, M. C. Dahlgreen, J. F. Franklin, P. F. Hessburg, and J. A. Lutz. 2013. Restoring forest resilience: From reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management 291:442–457.

Cogbill, C. V., A. L. Thurman, J. W. Williams, J. Zhu, D. J. Mladenoff, and S. J. Goring. 2018. A retrospective on the accuracy and precision of plotless forest density estimators in ecological studies. Ecosphere 9:e02187.

Collins, B. M., J. M. Lydersen, R. G. Everett, D. L. Fry, and S. L. Stephens. 2015. Novel characterization of landscape-level variability in historical vegetation structure. Ecological Applications 25:1167–1174.

Collins, B. M., J. M. Lydersen, R. G. Everett, and S. L. Stephens. 2018. How does forest recovery following moderate-severity fire influence effects of subsequent wildfire in mixed-conifer forests? Fire Ecology 14:3.

Collins, B. M., J. D. Miller, and S. L. Stephens. 2016. To the editor: a response to Hanson and Odion. Natural Areas Journal 36:234–242.

Collins, B. M., J. D. Miller, A. E. Thode, M. Kelly, J. W. van Wagtendonk, and S. L. Stephens. 2009. Interactions among wildland fires in a long-established Sierra Nevada natural fire area. Ecosystems 12:114–128.

Collins, B. M., and S. L. Stephens. 2007. Fire scarring patterns in Sierra Nevada wilderness areas burned by multiple wildland fire use fires. Fire Ecology 3:53–67.

Collins, B. M., J. T. Stevens, J. D. Miller, S. L. Stephens, P. M. Brown, and M. P. North. 2017. Alternative characterization of forest fire regimes: incorporating spatial patterns. Landscape Ecology 32:1543–1552.

Conver, J. L., D. A. Falk, S. R. Yool, and R. Parmenter. 2018. Stochastic fire modeling of a montane grassland and ponderosa pine fire regime in the Valles Caldera National Preserve, New Mexico, USA. Fire Ecology 14:17–31.

Coogan, S. C. P., L. D. Daniels, D. Boychuk, P. J. Burton, M. D. Flannigan, S. Gauthier, V. Kafka, J. S. Park, and B. M. Wotton. 2020. Fifty years of wildland fire science in Canada. Canadian Journal of Forest Research 51:283–302.

Coogan, S. C. P., F. Robinne, J. Piyush, and M. D. Flannigan. 2019. Scientists' warning on wildfire—a Canadian perspective. Canadian Journal of Forest Research 49:1015–1023.

Coop, J. D., et al. 2020. Wildfire-driven forest conversion in western North American landscapes. BioScience 70:659–673.

Coppoletta, M., K. E. Merriam, and B. M. Collins. 2016. Post-fire vegetation and fuel development influences fire severity patterns in reburns. Ecological Applications 26:686–699.

Coppoletta, M., M. D. Meyer, and M. P. North. 2021. Natural range of variation for red fir and subalpine forests in northwestern California and southwestern Oregon. General Technical Report PSW-GTR-269. US Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Cortés Montaño, C., P. Z. Fulé, D. A. Falk, J. Villanueva-Díaz, and L. L. Yocom. 2012. Linking old-growth forest composition, structure, fire history, climate and land-use in the mountains of northern México. Ecosphere 3:art106.

Daniels, L. D., and R. W. Gray. 2006. Disturbance regimes in coastal British Columbia. Journal of Ecosystems and Management 7:44–56.

Daniels, L. D., T. B. Maertens, A. B. Stan, S. P. J. McCloskey, J. D. Cochrane, and R. W. Gray. 2011. Direct and indirect impacts of climate change on forests: three case studies from British Columbia. Canadian Journal of Plant Pathology 33:108–116.

Daniels, L. D., L. L. Yocom Kent, R. L. Sherriff, and E. K. Heyerdahl. 2017. Deciphering the complexity of historical fire regimes: diversity among forests of western North America. Pages 185–210 *in* M. M. Amoroso, L. D. Daniels, P. J. Baker, and J. J. Camarero, editors. Dendroecology: tree-ring analyses applied to ecological studies. Springer International Publishing, Cham, Switzerland.

David, A. T., J. E. Asarian, and F. K. Lake. 2018. Wildfire smoke cools summer river and stream water temperatures. Water Resources Research 54:7273–7290.

Davis, K. T., S. Z. Dobrowski, P. E. Higuera, Z. A. Holden, T. T. Veblen, M. T. Rother, S. A. Parks, A. Sala, and M. P. Maneta. 2019. Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration. Proceedings of the National Academy of Sciences USA 116:6193–6198.

deBuys, W., and C. D. Allen. 2015. A historical chronology of events and observations for the Pecos wilderness in the territorial period. New Mexico Historical Review 90:415–487.

DellaSala, D. A., and C. T. Hanson. 2019. Are wildland fires increasing large patches of complex early seral forest habitat? Diversity 11:157.

Dewar, J. J., D. A. Falk, T. W. Swetnam, C. H. Baisan, C. D. Allen, R. R. Parmenter, E. Q. Margolis, and E. J. Taylor. 2021. Valleys of fire: historical fire regimes of forest-grassland ecotones across the montane landscape of the Valles Caldera National Preserve, New Mexico, USA. Landscape Ecology 36:331–352.

Diggins, C., P. Z. Fulé, J. P. Kaye, and W. W. Covington. 2010. Future climate affects management strategies for maintaining forest restoration treatments. International Journal of Wildland Fire 19:903–913.

Dillon, G. K., D. H. Knight, and C. B. Meyer. 2005. Historic range of variability for upland vegetation in the Medicine Bow National Forest, Wyoming. General Technical Report RMRS-GTR-139. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Dove, N. C., H. D. Safford, G. N. Bohlman, B. L. Estes, and S. C. Hart. 2020. High-severity wildfire leads to multi-decadal

impacts on soil biogeochemistry in mixed-conifer forests. Ecological Applications 30:e02072.

Dugan, A. J., and W. L. Baker. 2014. Modern calibration and historical testing of small-area, fire-interval reconstruction methods. International Journal of Wildland Fire 23:58–68.

Dumroese, R. K., and W. K. Moser. 2020. Northeastern California plateaus bioregion science synthesis. RMRS-GTR-409. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Ehle, D. S., and W. L. Baker. 2003. Disturbance and stand dynamics in ponderosa pine forests in Rocky Mountain National Park, USA. Ecological Monographs 73:543–566.

Eisenberg, C., et al. 2019. Out of the ashes: ecological resilience to extreme wildfire, prescribed burns, and indigenous burning in ecosystems. Frontiers in Ecology and Evolution 7:436.

Esch, B. E., A. E. M. Waltz, T. N. Wasserman, and E. L. Kalies. 2018. Using best available science information: determining best and available. Journal of Forestry 116:473–480.

Falk, D. A. 2013. Are Madrean ecosystems approaching tipping points? Anticipating interactions of landscape disturbance and climate change. Pages 40–47 *in* G. J. Gottfried, P. F. Ffolliott, B. S. Gebow, L. G. Eskew, and L. C. Collins, comps. 2013. Merging science and management in a rapidly changing world: Biodiversity and management of the Madrean Archipelago III; 2012 May 1-5; Tucson, AZ. Proceedings. RMRS-P-67. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Falk, D. A., E. K. Heyerdahl, P. M. Brown, C. Farris, P. Z. Fulé, D. McKenzie, T. W. Swetnam, A. H. Taylor, and M. L. V. Horne. 2011. Multi-scale controls of historical forest-fire regimes: new insights from fire-scar networks. Frontiers in Ecology and the Environment 9:446–454.

Falk, D. A., and T. W. Swetnam. 2003. Scaling rules and probability models for surface fire regimes in ponderosa pine forests. Pages 301–318 *in* P. N. Omi and L. A. Joyce, editors. Fire, Fuel Treatments and Ecological Restoration: Conference Proceedings: April 16-18, 2002. USDA Forest Service Proceedings RMRSP-29. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Falk, D. A., A. C. Watts, and A. E. Thode. 2019. Scaling ecological resilience. Frontiers in Ecology and Evolution 7:275.

Farris, C. A., C. H. Baisan, D. A. Falk, M. L. V. Horne, P. Z. Fulé, and T. W. Swetnam. 2013. A comparison of targeted and systematic fire-scar sampling for estimating historical fire frequency in south-western ponderosa pine forests. International Journal of Wildland Fire 22:1021–1033.

Farris, C. A., C. H. Baisan, D. A. Falk, S. R. Yool, and T. W. Swetnam. 2010. Spatial and temporal corroboration of a fire-scar-based fire history in a frequently burned ponderosa pine forest. Ecological Applications 20:1598–1614.

Fettig, C. J., L. A. Mortenson, B. M. Bulaon, and P. B. Foulk. 2019. Tree mortality following drought in the central and southern Sierra Nevada, California, US. Forest Ecology and Management 432:164–178.

Fettig, C. J., M. L. Reid, B. J. Bentz, S. Sevanto, D. L. Spittlehouse, and T. Wang. 2013. Changing climates, changing forests: a western North American perspective. Journal of Forestry 111:214–228.

Field, J., D. Breshears, J. Bradford, D. Law, X. Feng, and C. Allen. 2020. Forest management under megadrought: urgent needs at finer scale and higher intensity. Frontiers in Forests and Global Change 3:502669.

Fornwalt, P. J., L. S. Huckaby, S. K. Alton, M. R. Kaufmann, P. M. Brown, and A. S. Cheng. 2016. Did the 2002 Hayman Fire, Colorado, USA, burn with uncharacteristic severity? Fire Ecology 12:117–132.

Fortin, J. A., J. T. Fisher, J. M. Rhemtulla, and E. S. Higgs. 2019. Estimates of landscape composition from terrestrial oblique photographs suggest homogenization of Rocky Mountain landscapes over the last century. Remote Sensing in Ecology and Conservation 5:224–236.

Franklin, J. F., and J. K. Agee. 2003. Forging a science-based national forest fire policy. Issues in Science and Technology 20:59–66.

Franklin, J. F., K. N. Johnson, and D. L. Johnson. 2018. Ecological forest management. Waveland Press, Long Grove, Illinois, USA.

Franklin, J. F., D. B. Lindenmayer, J. A. MacMahon, A. McKee, J. Magnuson, D. A. Perry, R. Waide, and D. Foster. 2000. Threads of continuity. Conservation Biology in Practice 1:8–16.

Franklin, J. F., and R. Van Pelt. 2004. Spatial aspects of structural complexity in old-growth forests. Journal of Forestry 102:22–28.

Fry, D. L., S. L. Stephens, B. M. Collins, M. P. North, E. Franco-Vizcaíno, and S. J. Gill. 2014. Contrasting spatial patterns in active-fire and fire-suppressed Mediterranean climate old-growth mixed conifer forests. PLoS ONE 9:e88985.

Fulé, P. Z., J. E. Crouse, J. P. Roccaforte, and E. L. Kalies. 2012*a*. Do thinning and/or burning treatments in western USA ponderosa or Jeffrey pine-dominated forests help restore natural fire behavior? Forest Ecology and Management 269:68–81.

Fulé, P. Z., L. L. Yocom, C. C. Montaño, D. A. Falk, J. Cerano, and J. Villanueva-Díaz. 2012*b*. Testing a pyroclimatic hypothesis on the Mexico-United States border. Ecology 93:1830–1840.

Fulé, P. Z., T. A. Heinlein, W. W. Covington, and M. M. Moore. 2003. Assessing fire regimes on Grand Canyon landscapes with fire-scar and fire-record data. International Journal of Wildland Fire 12:129–145.

Fulé, P. Z., et al. 2014. Unsupported inferences of high-severity fire in historical dry forests of the western United States: response to Williams and Baker. Global Ecology and Biogeography 23:825–830.

Gartner, M. H., T. T. Veblen, R. L. Sherriff, and T. L. Schoennagel. 2012. Proximity to grasslands influences fire frequency and sensitivity to climate variability in ponderosa pine forests of the Colorado Front Range. International Journal of Wildland Fire 21:562–571.

Grafe, E., and P. Horsted. 2002. Exploring with custer: the 1874 Black Hills expedition. Golden Valley Press, Custer, South Dakota, USA.

Graham, R. T., A. E. Harvey, T. B. Jain, and J. R. Tonn. 1999. The effects of thinning and similar stand treatments on fire behavior in Western forests. PNW-GTR-463. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Graham, R. T., S. McCaffrey, and T. B. Jain. 2004. Science basis for changing forest structure to modify wildfire behavior and severity. RMRS-GTR-120. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Greene, G. A., and L. D. Daniels. 2017. Spatial interpolation and mean fire interval analyses quantify historical mixed-severity fire regimes. International Journal of Wildland Fire 26:136–147.

Grimm, V., and C. Wissel. 1997. Babel, or the ecological stability discussions: an inventory and analysis of terminology and a guide for avoiding confusion. Oecologia 109:323–334.

Guiterman, C. H., E. Q. Margolis, C. D. Allen, D. A. Falk, and T. W. Swetnam. 2018. Long-term persistence and fire resilience of oak shrubfields in dry conifer forests of northern New Mexico. Ecosystems 21:943–959.

Guiterman, C. H., E. Q. Margolis, and T. W. Swetnam. 2015. Dendroecological methods for reconstructing high-severity fire In pine-oak forests. Tree-Ring Research 71:67–77.

Hagmann, R. K., J. F. Franklin, and K. N. Johnson. 2013. Historical structure and composition of ponderosa pine and mixed-conifer forests in south-central Oregon. Forest Ecology and Management 304:492–504.

Hagmann, R. K., J. F. Franklin, and K. N. Johnson. 2014. Historical conditions in mixed-conifer forests on the eastern slopes of the northern Oregon Cascade Range, USA. Forest Ecology and Management 330:158–170.

Hagmann, R. K., D. L. Johnson, and K. N. Johnson. 2017. Historical and current forest conditions in the range of the Northern Spotted Owl in south central Oregon, USA. Forest Ecology and Management 389:374–385.

Hagmann, R. K., A. G. Merschel, and M. J. Reilly. 2019. Historical patterns of fire severity and forest structure and composition in a landscape structured by frequent large fires: Pumice Plateau ecoregion, Oregon, USA. Landscape Ecology 34:551–568.

Hagmann, R. K., et al. 2018. Improving the use of early timber inventories in reconstructing historical dry forests and fire in the western United States: Comment. Ecosphere 9:e02232.

Hamann, A., T. Wang, D. L. Spittlehouse, and T. Q. Murdock. 2013. A comprehensive, high-resolution database of historical and projected climate surfaces for western North America. Bulletin of the American Meteorological Society 94:1307–1309.

Hanberry, B. B., and R. K. Dumroese. 2020. Biodiversity and representative species in dry pine forests. Pages 85–100 *in* R. K. Dumroese and W. K. Moser, editors. Northeastern California plateaus bioregion science synthesis. RMRS-GTR-409. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Hanberry, B. B., S. Fraver, H. S. He, J. Yang, D. C. Dey, and B. J. Palik. 2011. Spatial pattern corrections and sample sizes for forest density estimates of historical tree surveys. Landscape Ecology 26:59–68.

Hann, W. J., et al. 1997. Chapter 3: Landscape dynamics of the basin. *In* T. M. Quigley and S. J. Arbelbide, editors. An assessment of ecosystem components in the interior Columbia basin and portions of the Klamath and Great Basins: Volume II. PNW-GTR-405. USDA Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Hann, W. J., J. L. Jones, R. E. Keane, P. F. Hessburg, and R. A. Gravenmier. 1998. ICBEMP: landscape dynamics. Journal of Forestry 96:10–15.

Hansen, A. J., T. A. Spies, F. J. Swanson, and J. L. Ohmann. 1991. Conserving biodiversity in managed forests. BioScience 41:382–392.

Hanson, C. T., and D. C. Odion. 2014. Is fire severity increasing in the Sierra Nevada, California, USA? International Journal of Wildland Fire 23:1–8.

Hanson, C. T., and D. C. Odion. 2016. Historical forest conditions within the range of the Pacific fisher and Spotted Owl in the central and southern Sierra Nevada, California, USA. Natural Areas Journal 36:8–19.

Hanson, C. T., D. C. Odion, D. A. Dellasala, and W. L. Baker. 2009. Overestimation of fire risk in the Northern Spotted Owl recovery plan. Conservation Biology 23:1314–1319.

Harris, L., and A. H. Taylor. 2015. Topography, fuels, and fire exclusion drive fire severity of the Rim Fire in an old-growth mixed-conifer forest, Yosemite National Park, USA. Ecosystems 18:1192–1208.

Harris, L. B., and A. H. Taylor. 2020. Rain-shadow forest margins resilient to low-severity fire and climate change but not high-severity fire. Ecosphere 11:e03258.

Haugo, R. D., B. S. Kellogg, C. A. Cansler, C. A. Kolden, K. B. Kemp, J. C. Robertson, K. L. Metlen, N. M. Vaillant, and C. M. Restaino. 2019. The missing fire: quantifying human exclusion of wildfire in Pacific Northwest forests, USA. Ecosphere 10:e02702.

He, H. S., R. E. Keane, and L. R. Iverson. 2008. Forest landscape models, a tool for understanding the effect of the large-scale and long-term landscape processes. Forest Ecology and Management 254:371–374, 254.

Hessburg, P. F., and J. K. Agee. 2003. An environmental narrative of Inland Northwest United States forests, 1800–2000. Forest Ecology and Management 178:23–59.

Hessburg, P. F., J. K. Agee, and J. F. Franklin. 2005. Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre-settlement and modern eras. Forest Ecology and Management 211:117–139.

Hessburg, P. F., et al. 2015. Restoring fire-prone Inland Pacific landscapes: seven core principles. Landscape Ecology 30:1805–1835.

Hessburg, P. F., et al. 2019. Climate, environment, and disturbance history govern resilience of western North American forests. Frontiers in Ecology and Evolution 7:239.

Hessburg, P. F., R. G. Mitchell, and G. M. Filip. 1994. Historical and current roles of insects and pathogens in eastern Oregon and Washington forested landscapes. General Technical Report PNW-GTR-327. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Hessburg, P. F., S. J. Prichard, R. K. Hagmann, N. A. Povak, and F. K. Lake. 2021. Wildfire and climate change adaptation of western North American forests: A case for intentional management. Ecological Applications, *in press.*

Hessburg, P. F., R. B. Salter, and K. M. James. 2007. Re-examining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. Landscape Ecology 22:5–24.

Hessburg, P. F., B. G. Smith, S. D. Kreiter, C. A. Miller, R. B. Salter, C. H. McNicoll, and W. J. Hann. 1999*a*. Historical and current forest and range landscapes in the interior Columbia River basin and portions of the Klamath and Great Basins. Part 1: Linking vegetation patterns and landscape vulnerability to potential insect and pathogen disturbances. PNW-GTR-458. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Hessburg, P. F., B. G. Smith, and R. B. Salter. 1999*b*. Detecting change in forest spatial patterns from reference conditions. Ecological Applications 9:1232–1252.

Hessburg, P. F., B. G. Smith, R. B. Salter, R. D. Ottmar, and E. Alvarado. 2000. Recent changes (1930s–1990s) in spatial patterns of interior northwest forests, USA. Forest Ecology and Management 136:53–83.

Hessburg, P. F., et al. 2016. Tamm Review: Management of mixed-severity fire regime forests in Oregon, Washington, and Northern California. Forest Ecology and Management 366:221–250.

Hessl, A., J. Miller, J. Kernan, D. Keenum, and D. McKenzie. 2007. Mapping paleo-fire boundaries from binary point data: comparing interpolation methods. Professional Geographer 59:87–104.

Heyerdahl, E. K., L. B. Brubaker, and J. K. Agee. 2001. Spatial controls of historical fire regimes: a multiscale example from the interior West, USA. Ecology 82:660–678.

Heyerdahl, E. K., R. A. Loehman, and D. A. Falk. 2014. Mixed-severity fire in lodgepole pine dominated forests: are historical regimes sustainable on Oregon's Pumice Plateau, USA? Canadian Journal of Forest Research 44:593–603.

Higgs, E., G. Bartley, and A. Fisher. 2009. The Mountain legacy project. Second edition. University of Victoria Press, Victoria, British Columbia, Canada.

Higgs, E., D. A. Falk, A. Guerrini, M. Hall, J. Harris, R. J. Hobbs, S. T. Jackson, J. M. Rhemtulla, and W. Throop. 2014. The changing role of history in restoration ecology. Frontiers in Ecology and the Environment 12:499–506.

Hoffman, K. M., K. P. Lertzman, and B. M. Starzomski. 2017. Ecological legacies of anthropogenic burning in a British Columbia coastal temperate rain forest. Journal of Biogeography 44:2903–2915.

Hoffman, K. M., S. B. Wickham, W. S. McInnes, and B. M. Starzomski. 2019. Fire exclusion destroys habitats for at-risk species in a British Columbia protected area. Fire 2:48.

Hogg, E. H. 1997. Temporal scaling of moisture and the forest-grassland boundary in western Canada. Agricultural and Forest Meteorology 84:115–122.

Holsinger, L., R. E. Keane, D. J. Isaak, L. Eby, and M. K. Young. 2014. Relative effects of climate change and wildfires on stream temperatures: a simulation modeling approach in a Rocky Mountain watershed. Climatic Change 124:191–206.

Hood, S. M., C. R. Keyes, K. J. Bowen, D. C. Lutes, and C. Seielstad 2020. Fuel treatment longevity in ponderosa pine-dominated forest 24 years after cutting and prescribed burning. Frontiers in Forests and Global Change 3:78.

Huff, M. H., R. D. Ottmar, E. Alvarado, R. E. Vihnanek, J. F. Lehmkuhl, P. F. Hessburg, and R. L. Everett. 1995. Historical and current forest landscapes of eastern Oregon and Washington: Part II: linking vegetation characteristics to potential fire behavior and related smoke production. PNW-GTR-355. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Huffman, D. W., T. J. Zegler, and P. Z. Fulé. 2015. Fire history of a mixed conifer forest on the Mogollon Rim, northern Arizona, USA. International Journal of Wildland Fire 24:680–689.

Jain, T. B., and R. T. Graham. 2007. The relation between tree burn severity and forest structure in the Rocky Mountains. Pages 213–250 *in* R. F. Powers, technical editor. Restoring Fire-Adapted Ecosystems: Proceedings of the 2005 National Silviculture Workshop. General Technical Report PSW-GTR-203. Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture, Albany, California, USA.

Johnson, L. B., and E. Q. Margolis. 2019. Surface fire to crown fire: fire history in the Taos Valley watersheds, New Mexico, USA. Fire 2:14.

Johnston, J. D. 2017. Forest succession along a productivity gradient following fire exclusion. Forest Ecology and Management 392:45–57.

Johnston, J. D., C. J. Dunn, M. J. Vernon, J. D. Bailey, B. A. Morrissette, and K. E. Morici. 2018. Restoring historical forest conditions in a diverse inland Pacific Northwest landscape. Ecosphere 9:e02400.

Johnstone, J. F., et al. 2016. Changing disturbance regimes, ecological memory, and forest resilience. Frontiers in Ecology and the Environment 14:369–378.

Jones, G. M., et al. 2020*a*. Spotted owls and forest fire: Comment. Ecosphere 11:e03312.

Jones, G. M., et al. 2020*b*. Habitat selection by spotted owls after a megafire reflects their adaptation to historical frequent-fire regimes. Landscape Ecology 35:1199–1213.

Jones, G. M. 2019. Megafire effects on spotted owls: elucidation of a growing threat and a response to Hanson et al. (2018). Nature Conservation 37:31–51.

Kane, V. R., B. N. Bartl-Geller, M. P. North, J. T. Kane, J. M. Lydersen, S. M. A. Jeronimo, B. M. Collins, and L. Monika Moskal. 2019. First-entry wildfires can create opening and tree clump patterns characteristic of resilient forests. Forest Ecology and Management 454:117659.

Kaufmann, M. R., D. Binkley, P. Z. Fulé, M. Johnson, S. L. Stephens, and T. W. Swetnam. 2007. Defining old growth for fire-adapted forests of the western United States. Ecology and Society 12:15.

Kaye, M. W., and T. W. Swetnam. 1999. An assessment of fire, climate, and Apache history in the Sacramento mountains, New Mexico. Physical Geography 20:305–330.

Keane, R. E. 1996. Simulating coarse-scale vegetation dynamics using the Columbia River basin succession model--CRBSUM. No. 339-340. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, Utah, USA.

Keane, R. E. 2019. Applying ecosystem and landscape models in natural resource management. CRC Press, Taylor and Francis Group, Boca Raton, Florida, USA.

Keane, R. E., P. F. Hessburg, P. B. Landres, and F. J. Swanson. 2009. The use of historical range and variability (HRV) in landscape management. Forest Ecology and Management 258:1025–1037.

Keane, R. E., R. A. Loehman, L. M. Holsinger, D. A. Falk, P. Higuera, S. M. Hood, and P. F. Hessburg. 2018. Use of landscape simulation modeling to quantify resilience for ecological applications. Ecosphere 9:e02414.

Keane, R. E., R. Parsons, and M. G. Rollins 2004. Predicting fire regimes across multiple scales. Pages 55–69 *in* A. Perera and L. Buse, editors. Emulating natural disturbances: concepts and techniques. Cambridge University Press, Cambridge, UK.

Keeley, J. E. 2012. Ecology and evolution of pine life histories. Annals of Forest Science 69:445–453.

King, K. J., R. A. Bradstock, G. J. Cary, J. Chapman, and J. B. Marsden-Smedley. 2008. The relative importance of fine-scale fuel mosaics on reducing fire risk in south-west Tasmania, Australia. International Journal of Wildland Fire 17:421–430.

Klimaszewski-Patterson, A., and S. A. Mensing. 2016. Multi-disciplinary approach to identifying Native American impacts on Late Holocene forest dynamics in the southern Sierra Nevada range, California, USA. Anthropocene 15:37–48.

Knapp, E. E., C. N. Skinner, M. P. North, and B. L. Estes. 2013. Long-term overstory and understory change following logging and fire exclusion in a Sierra Nevada mixed-conifer forest. Forest Ecology and Management 310:903–914.

Knight, C. A., C. V. Cogbill, M. D. Potts, J. A. Wanket, and J. J. Battles. 2020. Settlement-era forest structure and composition in the Klamath Mountains: reconstructing a historical baseline. Ecosphere 11:e03250.

Kolb, T. E., J. K. Agee, P. Z. Fulé, N. G. McDowell, K. Pearson, A. Sala, and R. H. Waring. 2007. Perpetuating old ponderosa pine. Forest Ecology and Management 249:141–157.

Kou, X., and W. L. Baker. 2006*a*. Accurate estimation of mean fire interval for managing fire. International Journal of Wildland Fire 15:489–495.

Kou, X., and W. L. Baker. 2006*b*. A landscape model quantifies error in reconstructing fire history from scars. Landscape Ecology 21:735–745.

Krawchuk, M. A., G. W. Meigs, J. M. Cartwright, J. D. Coop, R. Davis, A. Holz, C. Kolden, and A. J. Meddens. 2020. Disturbance refugia within mosaics of forest fire, drought, and insect outbreaks. Frontiers in Ecology and the Environment 18:235–244.

Lake, F. K., and J. W. Long. 2014. Fire and tribal cultural resources, Chapter 4.2. Pages 173–186 *in* J. W. Long, L. N. Quinn-Davidson, and C. N. Skinner, editors. Science synthesis to support socioecological resilience in the Sierra Nevada and southern Cascade Range. General Technical Report

PSW-GTR-247. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Lake, F. K., J. Parrotta, C. P. Giardina, I. Davidson-Hunt, and Y. Uprety. 2018. 12: Integration of traditional and western knowledge in forest landscape restoration. Pages 198–226 *in* S. Mansourian and J. Parrotta, editors. Forest landscape restoration: integrated approaches to support effective implementation. Routledge, New York, New York, USA.

Larson, A. J., and D. Churchill. 2012. Tree spatial patterns in fire-frequent forests of western North America, including mechanisms of pattern formation and implications for designing fuel reduction and restoration treatments. Forest Ecology and Management 267:74–92.

Laughlin, D. C., R. T. Strahan, D. W. Huffman, and A. J. S. Meador. 2017. Using trait-based ecology to restore resilient ecosystems: historical conditions and the future of montane forests in western North America. Restoration Ecology 25: S135–S146.

Lee, D. E. 2018. Spotted Owls and forest fire: a systematic review and meta-analysis of the evidence. Ecosphere 9: e02354.

LeFevre, M. E., D. J. Churchill, A. J. Larson, S. M. A. Jeronimo, J. Bass, J. F. Franklin, and V. R. Kane. 2020. Evaluating restoration treatment effectiveness through a comparison of residual composition, structure, and spatial pattern with historical reference sites. Forest Science 66:578–588.

Lehmkuhl, J. F., P. F. Hessburg, R. L. Everett, M. H. Huff, and R. D. Ottmar. 1994. Historical and current forest landscapes of eastern Oregon and Washington: Part I: vegetation pattern and insect and EISEASE hazards. PNW-GTR-328. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Levine, C. R., C. V. Cogbill, B. M. Collins, A. J. Larson, J. A. Lutz, M. P. North, C. M. Restaino, H. D. Safford, S. L. Stephens, and J. J. Battles. 2017. Evaluating a new method for reconstructing forest conditions from General Land Office survey records. Ecological Applications 27:1498–1513.

Levine, C. R., C. V. Cogbill, B. M. Collins, A. J. Larson, J. A. Lutz, M. P. North, C. M. Restaino, H. D. Safford, S. L. Stephens, and J. J. Battles. 2019. Estimating historical forest density from land-survey data: A response to Baker and Williams (2018). Ecological Applications 29:e01968.

Loehman, R., W. Flatley, L. Holsinger, and A. Thode. 2018. Can land management buffer impacts of climate changes and altered fire regimes on ecosystems of the Southwestern United States? Forests 9:192.

Loehman, R. A., R. E. Keane, and L. M. Holsinger. 2020. Simulation modeling of complex climate, wildfire, and vegetation dynamics to address wicked problems in land management. Frontiers in Forests and Global Change 3:Article 3.

Loehman, R. A., R. E. Keane, L. M. Holsinger, and Z. Wu. 2017. Interactions of landscape disturbances and climate change dictate ecological pattern and process: spatial modeling of wildfire, insect, and disease dynamics under future climates. Landscape Ecology 32:1447–1459.

Long, J. W., M. K. Anderson, L. Quinn-Davidson, R. W. Goode, F. K. Lake, and C. N. Skinner. 2015. Restoring California black oak ecosystems to promote tribal values and wildlife. General Technical Report PSW GTR-252. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Long, J. W., A. Gray, and F. K. Lake. 2018. Recent trends in large hardwoods in the Pacific Northwest, USA. Forests 9:651.

Long, J. W., and F. K. Lake. 2018. Escaping social-ecological traps through tribal stewardship on national forest lands in the Pacific Northwest, United States of America. Ecology and Society 23:10.

Long, J. W., F. K. Lake, R. W. Goode, and B. M. Burnette. 2020. How traditional tribal perspectives influence ecosystem restoration. Ecopsychology 12:71–82.

Lydersen, J. M., and B. M. Collins. 2018. Change in vegetation patterns over a large forested landscape based on historical and contemporary aerial photography. Ecosystems 21:1348–1363.

Lydersen, J. M., B. M. Collins, M. L. Brooks, J. R. Matchett, K. L. Shive, N. A. Povak, V. R. Kane, and D. F. Smith. 2017. Evidence of fuels management and fire weather influencing fire severity in an extreme fire event. Ecological Applications 27:2013–2030.

Lydersen, J., and M. North. 2012. Topographic variation in structure of mixed-conifer forests under an active-fire regime. Ecosystems 15:1134–1146.

Lydersen, J. M., M. P. North, E. E. Knapp, and B. M. Collins. 2013. Quantifying spatial patterns of tree groups and gaps in mixed-conifer forests: reference conditions and long-term changes following fire suppression and logging. Forest Ecology and Management 304:370–382.

Ma, Q., R. C. Bales, J. Rungee, M. H. Conklin, B. M. Collins, and M. L. Goulden. 2020. Wildfire controls on evapotranspiration in California's Sierra Nevada. Journal of Hydrology 590:125364.

Maier, C., and J. B. Abrams. 2018. Navigating social forestry—A street-level perspective on National Forest management in the US Pacific Northwest. Land Use Policy 70:432–441.

Mallek, C., H. Safford, J. Viers, and J. Miller. 2013. Modern departures in fire severity and area vary by forest type, Sierra Nevada and southern Cascades, California, USA. Ecosphere 4:1–28.

Marcoux, H. M., L. D. Daniels, S. E. Gergel, E. Da Silva, Z. Gedalof, and P. F. Hessburg. 2015. Differentiating mixed- and high-severity fire regimes in mixed-conifer forests of the Canadian Cordillera. Forest Ecology and Management 341:45–58.

Margolis, E. Q., and S. B. Malevich. 2016. Historical dominance of low-severity fire in dry and wet mixed-conifer forest habitats of the endangered terrestrial Jemez Mountains salamander (*Plethodon neomexicanus*). Forest Ecology and Management 375:12–26.

Marks-Block, T., F. K. Lake, R. Bliege Bird, and L. M. Curran. 2021. Revitalized Karuk and Yurok cultural burning to enhance California hazelnut for basketweaving in northwestern California, USA. Fire Ecology 17:6.

Marlon, J. R., et al. 2012. Long-term perspective on wildfires in the western USA. Proceedings of the National Academy of Sciences USA 109:E535–E543.

Matonis, M. S., and D. Binkley. 2018. Not just about the trees: Key role of mosaic-meadows in restoration of ponderosa pine ecosystems. Forest Ecology and Management 411:120–131.

McCord, M. G., M. J. Reilly, E. S. Jules, and R. J. Butz. 2020. Early seral pathways of vegetation change following repeated, short interval high-severity wildfire in a low elevation mixed-conifer-hardwood forest landscape of the Klamath Mountains, California. Canadian Journal of Forest Research 50:13–23.

McIntyre, P. J., J. H. Thorne, C. R. Dolanc, A. L. Flint, L. E. Flint, M. Kelly, and D. D. Ackerly. 2015. Twentieth-century shifts in forest structure in California: Denser forests, smaller trees, and increased dominance of oaks. Proceedings of the National Academy of Sciences USA 112:1458–1463.

McKelvey, K. S., and J. D. Johnston. 1992. Historical perspectives on forests of the Sierra Nevada and the transverse ranges of southern California; forest conditions at the turn of

the century. Page 225–246 *in* J. Verner, K. S. McKelvey, B. R. Noon, R. J. Gutierrez, Jr., G. I. Gould, and T. W. Beck, Technical Coordinators. The California spotted owl: a technical assessment of its current status. General Technical Report PSW-GTR-133. Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture, Albany, California, USA.

McKinney, S. T. 2019. Systematic review and meta-analysis of fire regime research in ponderosa pine (*Pinus ponderosa*) ecosystems, Colorado, USA. Fire Ecology 15:38.

Merschel, A. G., P. A. Beedlow, D. C. Shaw, D. R. Woodruff, H. E. Lee, S. P. Cline, R. L. Comeleo, R. K. Hagmann, and M. J. Reilly. 2021. An ecological perspective on living with fire in ponderosa pine forests of Oregon and Washington: resistance, gone but not forgotten. Trees, Forests and People 4:100074.

Merschel, A. G., T. A. Spies, and E. K. Heyerdahl. 2014. Mixed-conifer forests of central Oregon: effects of logging and fire exclusion vary with environment. Ecological Applications 24:1670–1688.

Metlen, K. L., C. N. Skinner, D. R. Olson, C. Nichols, and D. Borgias. 2018. Regional and local controls on historical fire regimes of dry forests and woodlands in the Rogue River Basin, Oregon, USA. Forest Ecology and Management 430:43–58.

Meunier, J., N. S. Holoubek, and M. Sebasky. 2019. Fire regime characteristics in relation to physiography at local and landscape scales in Lake States pine forests. Forest Ecology and Management 454:117651.

Meyer, C. B., D. H. Knight, and G. K. Dillon. 2005*a*. Historic range of variability for upland vegetation in the Bighorn National Forest, Wyoming. General Technical Report RMRS-GTR-140. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Meyer, C. B., D. H. Knight, and G. K. Dillon. 2005*b*. Historic variability for the upland vegetation of the Shoshone National Forest, Wyoming. Report submitted to the Shoshone National Forest, supervisor's office, Cody, Wyoming and the division of renewable resources, Rocky Mountain region, US Forest Service, Lakewood, Colorado, USA.

Meyer, M. D., B. L. Estes, A. Wuenschel, B. Bulaon, A. Stucy, D. F. Smith, and A. C. Caprio. 2019. Structure, diversity and health of Sierra Nevada red fir forests with reestablished fire regimes. International Journal of Wildland Fire 28:386.

Meyer, M., and M. North. 2019. Natural range of variation of red fir and subalpine forests in the Sierra Nevada Bioregion. General Technical Report PSW-GTR-263. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Meyer, M. D., M. P. North, A. N. Gray, and H. S. J. Zald. 2007. Influence of soil thickness on stand characteristics in a Sierra Nevada mixed-conifer forest. Plant and Soil 294:113–123.

Miller, J. D., E. E. Knapp, C. H. Key, C. N. Skinner, C. J. Isbell, R. M. Creasy, and J. W. Sherlock. 2009*a*. Calibration and validation of the relative differenced Normalized Burn Ratio (RdNBR) to three measures of fire severity in the Sierra Nevada and Klamath Mountains, California, USA. Remote Sensing of Environment 113:645–656.

Miller, J. D., and B. Quayle. 2015. Calibration and validation of immediate post-fire satellite-derived data to three severity metrics. Fire Ecology 11:12–30.

Miller, J. D., and H. D. Safford. 2008. Sierra Nevada fire severity monitoring: 1984–2004. Tech. Report R5-TP-027. U.S. Department of Agriculture, Forest Service, Pacific Southwest Region, Vallejo, California, USA, 102 p.

Miller, J. D., and H. D. Safford. 2017. Corroborating evidence of a pre-Euro-American low- to moderate-severity fire regime in Yellow pine-mixed conifer forests of the Sierra Nevada, California, USA. Fire Ecology 13:58–90.

Miller, J. D., H. D. Safford, M. Crimmins, and A. E. Thode. 2009*b*. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. Ecosystems 12:16–32.

Moreira, F., et al. 2020. Wildfire management in Mediterranean-type regions: paradigm change needed. Environmental Research Letters 15:011001.

Morgan, P., G. H. Aplet, J. B. Haufler, H. C. Humphries, M. M. Moore, and W. D. Wilson. 1994. Historical range of variability. Journal of Sustainable Forestry 2:87–111.

Morgan, P., R. E. Keane, G. K. Dillon, T. B. Jain, A. T. Hudak, E. C. Karau, P. G. Sikkink, Z. A. Holden, and E. K. Strand. 2014. Challenges of assessing fire and burn severity using field measures, remote sensing and modelling. International Journal of Wildland Fire 23:1045–1060.

Morisita, M. 1957. A new method for the estimation of density by the spacing method applicable to nonrandomly distributed populations. Physiology and Ecology 7:134–144. English language translation available from http://math.hws.edu/~mitchell/Morisita1957.pdf

Moritz, M. A., C. Topik, C. D. Allen, P. F. Hessburg, P. Morgan, D. C. Odion, T. T. Veblen, and I. M. McCullough. 2018. A statement of common ground regarding the role of wildfire in forested landscapes of the western United States. Fire research consensus working group final report. http://www.nwfirescience.org/biblio/statement-common-ground-regarding-role-wildfire-forested-landscapes-western-united-states

Morrow, R. J. 1985. Age structure and spatial pattern of old-growth ponderosa pine in Pringle Falls Experimental Forest, central Oregon. Dissertation. Oregon State University, Corvallis, Oregon, USA.

Mueller, S. E., A. E. Thode, E. Q. Margolis, L. L. Yocom, J. D. Young, and J. M. Iniguez. 2020. Climate relationships with increasing wildfire in the southwestern US from 1984 to 2015. Forest Ecology and Management 460:117861.

Murphy, J. S., R. York, H. Rivera Huerta, and S. L. Stephens. 2021. Characteristics and metrics of resilient forests in the Sierra de San Pedro Martír, Mexico. Forest Ecology and Management 482:118864.

Naficy, C. E. 2017. A cross-scale assessment of historical fire severity patterns, landscape dynamics, and methodological challenges in mixed-severity fire regimes of the northern US Rockies. Dissertation. University of Colorado, Boulder, Colorado, USA.

Naficy, C., A. Sala, E. G. Keeling, J. Graham, and T. H. DeLuca. 2010. Interactive effects of historical logging and fire exclusion on ponderosa pine forest structure in the northern Rockies. Ecological Applications 20:1851–1864.

Ng, J., M. P. North, A. J. Arditti, M. R. Cooper, and J. A. Lutz. 2020. Topographic variation in tree group and gap structure in Sierra Nevada mixed-conifer forests with active fire regimes. Forest Ecology and Management 472:118220.

Nigro, K., and N. Molinari. 2019. Status and trends of fire activity in southern California yellow pine and mixed conifer forests. Forest Ecology and Management 441:20–31.

Norgaard, K. M. 2014. The politics of fire and the social impacts of fire exclusion on the Klamath. Humboldt Journal of Social Relations 36:77–101.

North, M. P., B. M. Collins, H. D. Safford, and N. L. Stephenson. 2016. Montane forests. Pages 553–577 *in* H. A. Mooney and E. Zavaleta, editors. Ecosystems of California. University of California Press, Berkeley, California, USA.

North, M., M. Hurteau, R. Fiegener, and M. Barbour. 2005*a*. Influence of fire and El Niño on tree recruitment varies by species in Sierran Mixed Conifer. Forest Science 51:187–197.

North, M., B. Oakley, R. Fiegener, A. Gray, and M. Barbour. 2005*b*. Influence of light and soil moisture on Sierran mixed-conifer understory communities. Plant Ecology 177:13–24.

North, M., et al. 2002. Vegetation and ecological characteristics of mixed-conifer and red fir forests at the Teakettle Experimental Forest. Technical Report PSW-GTR-186. Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture, Albany, California, USA.

North, M. P., S. L. Stephens, B. M. Collins, J. K. Agee, G. Aplet, J. F. Franklin, and P. Z. Fulé. 2015. Reform forest fire management. Science 349:1280–1281.

North, M., P. Stine, K. O'Hara, W. Zielinski, and S. Stephens. 2009. An ecosystem management strategy for Sierran mixed-conifer forests. General Technical Report PSW-GTR-220 (Second printing, with addendum). U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Noss, R. F., J. F. Franklin, W. L. Baker, T. Schoennagel, and P. B. Moyle. 2006. Managing fire-prone forests in the western United States. Frontiers in Ecology and the Environment 4:481–487.

O'Connor, C. D., D. A. Falk, A. M. Lynch, and T. W. Swetnam. 2014*a*. Fire severity, size, and climate associations diverge from historical precedent along an ecological gradient in the Pinaleño Mountains, Arizona, USA. Forest Ecology and Management 329:264–278.

O'Connor, C. D., D. A. Falk, A. M. Lynch, and T. W. Swetnam. 2014*b*. Fire severity, size, and climate associations diverge from historical precedent along an ecological gradient in the Pinaleño Mountains, Arizona, USA. Forest Ecology and Management 329:264–278.

O'Connor, C. D., D. A. Falk, A. M. Lynch, T. W. Swetnam, and C. P. Wilcox. 2017. Disturbance and productivity interactions mediate stability of forest composition and structure. Ecological Applications 27:900–915.

Odion, D. C., and C. T. Hanson. 2006. Fire severity in conifer forests of the Sierra Nevada, California. Ecosystems 9:1177–1189.

Odion, D. C., et al. 2014. Examining historical and current mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. PLoS ONE 9:e87852.

Parks, S. A., and J. T. Abatzoglou. 2020. Warmer and drier fire seasons contribute to increases in area burned at high severity in western US forests from 1985–2017. Geophysical Research Letters:e2020GL089858.

Parks, S. A., L. M. Holsinger, M. H. Panunto, W. M. Jolly, S. Z. Dobrowski, and G. K. Dillon. 2018. High-severity fire: evaluating its key drivers and mapping its probability across western US forests. Environmental Research Letters 13:044037.

Parks, S. A., C. Miller, C. R. Nelson, and Z. A. Holden. 2014. Previous fires moderate burn severity of subsequent wildland fires in two large western US wilderness areas. Ecosystems 17:29–42.

Parks, S. A., C. Miller, M.-A. Parisien, L. M. Holsinger, S. Z. Dobrowski, and J. Abatzoglou. 2015. Wildland fire deficit and surplus in the western United States, 1984–2012. Ecosphere 6:1–13.

Pawlikowski, N. C., M. Coppoletta, E. Knapp, and A. H. Taylor. 2019. Spatial dynamics of tree group and gap structure in an old-growth ponderosa pine-California black oak forest burned by repeated wildfires. Forest Ecology and Management 434:289–302.

Peery, M. Z., G. M. Jones, R. J. Gutierrez, S. M. Redpath, A. B. Franklin, D. Simberloff, M. G. Turner, V. C. Radeloff, and G. C. White. 2019. The conundrum of agenda-driven science in conservation. Frontiers in Ecology and the Environment 17:80–82.

Pellatt, M. G., and Z. Gedalof. 2014. Environmental change in Garry oak (*Quercus garryana*) ecosystems: the evolution of an eco-cultural landscape. Biodiversity and Conservation 23:2053–2067.

Perry, D. A., P. F. Hessburg, C. N. Skinner, T. A. Spies, S. L. Stephens, A. H. Taylor, J. F. Franklin, B. McComb, and G. Riegel. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. Forest Ecology and Management 262:703–717.

Povak, N. A., V. R. Kane, B. M. Collins, J. M. Lydersen, and J. T. Kane. 2020. Multi-scaled drivers of severity patterns vary across land ownerships for the 2013 Rim Fire, California. Landscape Ecology 35:293–318.

Prichard, S. J., P. F. Hessburg, R. W. Gray, N. A. Povak, R. B. Salter, C. Stevens-Rumann & P. Morgan. 2018. Final Report: evaluating the influence of prior burn mosaics on subsequent wildfire behavior, severity, and fire management options. Joint Fire Science Project 14_1_02_30. https://www.firescience.gov/projects/14-1-02-30/project/14-1-02-30_final_report.pdf

Prichard, S. J., et al. 2021. Adapting western US forests to wildfires and climate change: 10 key questions. Ecological Applications, *in press*.

Prichard, S. J., C. S. Stevens-Rumann, and P. F. Hessburg. 2017. Tamm Review: Shifting global fire regimes: Lessons from reburns and research needs. Forest Ecology and Management 396:217–233.

Rakhmatulina, E., G. Boisramé, S. L. Stephens, and S. Thompson. 2021. Hydrological benefits of restoring wildfire regimes in the Sierra Nevada persist in a warming climate. Journal of Hydrology 593:125808.

Raphael, M. G., M. J. Wisdom, M. M. Rowland, R. S. Holthausen, B. C. Wales, B. G. Marcot, and T. D. Rich. 2001. Status and trends of habitats of terrestrial vertebrates in relation to land management in the interior Columbia river basin. Forest Ecology and Management 153:63–87.

Reilly, M. J., C. J. Dunn, G. W. Meigs, T. A. Spies, R. E. Kennedy, J. D. Bailey, and K. Briggs. 2017. Contemporary patterns of fire extent and severity in forests of the Pacific Northwest, USA (1985–2010). Ecosphere 8:e01695.

Reilly, M. J., M. Elia, T. A. Spies, M. J. Gregory, G. Sanesi, and R. Lafortezza. 2018. Cumulative effects of wildfires on forest dynamics in the eastern Cascade Mountains, USA. Ecological Applications 28:291–308.

Reilly, M. J., V. J. Monleon, E. S. Jules, and R. J. Butz. 2019. Range-wide population structure and dynamics of a serotinous conifer, knobcone pine (*Pinus attenuata* L.), under an anthropogenically-altered disturbance regime. Forest Ecology and Management 441:182–191.

Restaino, C., D. J. N. Young, B. Estes, S. Gross, A. Wuenschel, M. Meyer, and H. Safford. 2019. Forest structure and climate mediate drought-induced tree mortality in forests of the Sierra Nevada, USA. Ecological Applications 29:e01902.

Reynolds, R. T., A. J. S. Meador, J. A. Youtz, T. Nicolet, M. S. Matonis, P. L. Jackson, D. G. DeLorenzo, and A. D. Graves. 2013. Restoring composition and structure in Southwestern frequent-fire forests: A science-based framework for improving ecosystem resiliency. General Technical Report RMRS-GTR-310. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Rhemtulla, J. M., R. J. Hall, E. S. Higgs, and S. E. Macdonald. 2011. Eighty years of change: vegetation in the montane

ecoregion of Jasper National Park, Alberta, Canada. Canadian Journal of Forest Research 32:2010–2021.

Riegel, G. N., R. F. Miller, C. N. Skinner, S. E. Smith, C. A. Farris, and K. E. Merriam. 2018. Northeastern plateaus bioregion. Pages 229–248 *in* J. W. van Wagtendonk, N. G. Sugihara, S. L. Stephens, A. E. Thode, K. E. Shaffer, and J. Fites- Kaufman, editors. Fire in California's ecosystems. Second edition. University of California Press, Berkeley, California, USA.

Riggs, R. A., et al. 2015. Biomass and fire dynamics in a temperate forest-grassland mosaic: Integrating multi-species herbivory, climate, and fire with the FireBGCv2/GrazeBGC system. Ecological Modelling 296:57–78.

Rivera-Huerta, H., H. D. Safford, and J. D. Miller. 2016. Patterns and trends in burned area and fire severity from 1984 to 2010 in the Sierra de San Pedro Mártir, Baja California, Mexico. Fire Ecology 12:52–72.

Roccaforte, J. P., P. Z. Fulé, and W. W. Covington. 2008. Landscape-scale changes in canopy fuels and potential fire behaviour following ponderosa pine restoration treatments. International Journal of Wildland Fire 17:293–303.

Rogeau, M.-P., M.-A. Parisien, and M. D. Flannigan. 2016. Fire history sampling strategy of fire intervals associated with mixed- to full-severity fires in southern Alberta, Canada. Forest Science 62:613–622.

Rollins, M. G. 2009. LANDFIRE: a nationally consistent vegetation, wildland fire, and fuel assessment. International Journal of Wildland Fire 18:235–249.

Roos, C. I., et al. 2021. Native American fire management at an ancient wildland–urban interface in the Southwest United States. Proceedings of the National Academy of Sciences USA 118:e2018733118.

Rummell, R. S. 1951. Some effects of livestock grazing on ponderosa pine forest and range in central Washington. Ecology 32:594–607.

Safford, H. D., J. D. Miller, and B. M. Collins. 2015. Differences in land ownership, fire management objectives and source data matter: a reply to Hanson and Odion (2014). International Journal of Wildland Fire 24:286–293.

Safford, H. D., J. Miller, D. Schmidt, B. Roath, and A. Parsons. 2008. BAER soil burn severity maps do not measure fire effects to vegetation: A comment on Odion and Hanson (2006). Ecosystems 11:1–11.

Safford, H. D., and J. T. Stevens. 2017. Natural range of variation for yellow pine and mixed-conifer forests in the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests, California, USA. General Technical Report PSW-GTR-256. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Savage, M., and T. W. Swetnam. 1990. Early 19th-century fire decline following sheep pasturing in a Navajo ponderosa pine forest. Ecology 71:2374–2378.

Schneider, E. E., D. L. R. Affleck, and A. J. Larson. 2019. Tree spatial patterns modulate peak snow accumulation and snow disappearance. Forest Ecology and Management 441:9–19.

Scholl, A. E., and A. H. Taylor. 2010. Fire regimes, forest change, and self-organization in an old-growth mixed-conifer forest, Yosemite National Park, USA. Ecological Applications 20:362–380.

Shinneman, D. J., and W. L. Baker. 1997. Nonequilibrium dynamics between catastrophic disturbances and old-growth forests in Ponderosa pine landscapes of the Black Hills. Conservation Biology 11:1276–1288.

Shive, K. L., H. K. Preisler, K. R. Welch, H. D. Safford, R. J. Butz, K. L. O'Hara, and S. L. Stephens. 2018. From the stand scale to the landscape scale: predicting the spatial patterns of forest regeneration after disturbance. Ecological Applications 28:1626–1639.

Simpson, K. J., B. S. Ripley, P.-A. Christin, C. M. Belcher, C. E. R. Lehmann, G. H. Thomas, and C. P. Osborne. 2016. Determinants of flammability in savanna grass species. Journal of Ecology 104:138–148.

Singer, J. A., R. Turnbull, M. Foster, C. Bettigole, B. R. Frey, M. C. Downey, K. R. Covey, and M. S. Ashton. 2019. Sudden aspen decline: a review of pattern and process in a changing climate. Forests 10:671.

Skinner, C. N., A. H. Taylor, J. K. Agee, C. E. Briles, and C. L. Whitlock. 2018. Klamath mountains bioregion. Pages 171–194 *in* J. W. van Wagtendonk, N. G. Sugihara, S. L. Stephens, A. E. Thode, K. E. Shaffer, and J. Fites-Kaufman, editors. Fire in California's ecosystems. Second edition. University of California Press, Berkeley, California, USA.

SNEP. 1996. Sierra Nevada Ecosystem Project final report to Congress: Status of the Sierra Nevada. Davis, Centers for Water and Wildland Resources, University of California, Davis, California, USA.

Sowerwine, J., M. Mucioki, D. Sarna-Wojcicki, and L. Hillman. 2019. Reframing food security by and for Native American communities: a case study among tribes in the Klamath River basin of Oregon and California. Food Security 11:579–607.

Spies, T. A., et al. 2017. Using an agent-based model to examine forest management outcomes in a fire-prone landscape in Oregon, USA. Ecology and Society 22:25.

Spies, T. A., M. A. Hemstrom, A. Youngblood, and S. Hummel. 2006. Conserving old-growth forest diversity in disturbance-prone landscapes. Conservation Biology 20:351–362.

Spies, T. A., P. F. Hessburg, C. N. Skinner, K. J. Puettmann, M. J. Reilly, R. J. Davis, J. A. Kertis, J. W. Long, and D. C. Shaw. 2018*a*. Chapter 3: Old growth, disturbance, forest succession, and management in the area of the Northwest Forest Plan. Page 95–243 *in* T. A. Spies, P. A. Stine, R. Gravenmier, J. W. Long, and M. J. Reilly, technical coordinators. Synthesis of science to inform land management within the Northwest Forest Plan area. PNW-GTR-966. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Spies, T. A., P. A. Stine, R. A. Gravenmier, J. W. Long, and M. J. Reilly. 2018*b*. Volume 1—Synthesis of science to inform land management within the Northwest forest plan area. PNW-GTR-966. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Spies, T. A., et al. 2019. Twenty-five years of the Northwest Forest Plan: what have we learned? Frontiers in Ecology and the Environment 17:511–520.

Spies, T. A. 2010. Underestimating risks to the northern spotted owl in fire-prone forests: response to Hanson et al. Conservation Biology 24:330–333.

Steel, Z. L., M. J. Koontz, and H. D. Safford. 2018. The changing landscape of wildfire: burn pattern trends and implications for California's yellow pine and mixed conifer forests. Landscape Ecology 33:1159–1176.

Steel, Z. L., H. D. Safford, and J. H. Viers. 2015. The fire frequency-severity relationship and the legacy of fire suppression in California forests. Ecosphere 6:art8.

Stephens, S. L., B. M. Collins, E. Biber, and P. Z. Fulé. 2016. U.S. federal fire and forest policy: emphasizing resilience in dry forests. Ecosphere 7:e01584.

Stephens, S. L., B. M. Collins, C. J. Fettig, M. A. Finney, C. M. Hoffman, E. E. Knapp, M. P. North, H. Safford, and R. B. Wayman. 2018*a*. Drought, tree mortality, and wildfire in forests adapted to frequent fire. BioScience 68:77–88.

Stephens, S. L., J. W. Kanem, J. D. Stuart. 2018*b*. North coast bioregion. Pages 149–170 *in* J. Van Wagtendonk, editor. Fire in California's ecosystems. Second edition. University of California Press, Berkeley, California, USA.

Stephens, S. L., D. L. Fry, B. M. Collins, C. N. Skinner, E. Franco-Vizcaíno, and T. J. Freed. 2010. Fire-scar formation in Jeffrey pine—mixed conifer forests in the Sierra San Pedro Mártir, Mexico. Canadian Journal of Forest Research 40:1497–1505.

Stephens, S. L., and P. Z. Fulé. 2005. Western pine forests with continuing frequent fire regimes: Possible reference sites for management. Journal of Forestry 103:357–362.

Stephens, S. L., et al. 2019. Is fire "for the birds"? How two rare species influence fire management across the US. Frontiers in Ecology and the Environment 17:391–399.

Stephens, S. L., J. M. Lydersen, B. M. Collins, D. L. Fry, and M. D. Meyer. 2015. Historical and current landscape-scale ponderosa pine and mixed conifer forest structure in the Southern Sierra Nevada. Ecosphere 6:1–63.

Stephens, S. L., J. T. Stevens, B. M. Collins, R. A. York, and J. M. Lydersen. 2018*c*. Historical and modern landscape forest structure in fir (Abies)-dominated mixed conifer forests in the northern Sierra Nevada, USA. Fire Ecology 14:1–14.

Stephens, S. L., R. E. Martin, and N. E. Clinton. 2007. Prehistoric fire area and emissions from California's forests, woodlands, shrublands, and grasslands. Forest Ecology and Management 251:205–216.

Stephens, S. L., A. L. Westerling, M. D. Hurteau, M. Z. Peery, C. A. Schultz, and S. Thompson. 2021. Undesirable outcomes in seasonally dry forests. Frontiers in Ecology and the Environment 19:87–88.

Stephens, S. L., A. L. Westerling, M. D. Hurteau, Z. M. Peery, C. A. Schultz, and S. Thompson. 2020. Fire and climate change: conserving seasonally dry forests is still possible. Frontiers in Ecology and the Environment 18:354–360.

Stevens, J. T., B. M. Collins, J. D. Miller, M. P. North, and S. L. Stephens. 2017. Changing spatial patterns of stand-replacing fire in California conifer forests. Forest Ecology and Management 406:28–36.

Stevens, J. T., M. M. Kling, D. W. Schwilk, J. M. Varner, and J. M. Kane. 2020. Biogeography of fire regimes in western U.S. conifer forests: a trait-based approach. Global Ecology and Biogeography 29:944–955.

Stevens, J. T., et al. 2016. Average stand age from forest inventory plots does not describe historical fire regimes in ponderosa pine and mixed-conifer forests of western North America. PLoS ONE 11:e0147688.

Stevens-Rumann, C. S., K. B. Kemp, P. E. Higuera, B. J. Harvey, M. T. Rother, D. C. Donato, P. Morgan, and T. T. Veblen. 2018. Evidence for declining forest resilience to wildfires under climate change. Ecology Letters 21:243–252.

Stevens-Rumann, C. S., and P. Morgan. 2019. Tree regeneration following wildfires in the western US: a review. Fire Ecology 15:15.

Stevens-Rumann, C. S., S. J. Prichard, E. K. Strand, and P. Morgan. 2016. Prior wildfires influence burn severity of subsequent large fires. Canadian Journal of Forest Research 46:1375–1385.

Stine, P., et al. 2014. The ecology and management of moist mixed-conifer forests in eastern Oregon and Washington: a synthesis of the relevant biophysical science and implications for future land management. PNW-GTR-897. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Stockdale, C. A., S. E. Macdonald, and E. Higgs. 2019*a*. Forest closure and encroachment at the grassland interface: a century-scale analysis using oblique repeat photography. Ecosphere 10:e02774.

Stockdale, C. A., N. McLoughlin, M. Flannigan, and S. E. Macdonald. 2019*b*. Could restoration of a landscape to a pre-European historical vegetation condition reduce burn probability? Ecosphere 10:e02584.

Stokes, M. A., and J. H. Dieterich. 1980. Proceedings of the fire history workshop. October 20-24, 1980, Tucson, Arizona. RM-GTR-81. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado, USA.

Strahan, R. T., A. J. S. Meador, D. W. Huffman, and D. C. Laughlin. 2016. Shifts in community-level traits and functional diversity in a mixed conifer forest: a legacy of land-use change. Journal of Applied Ecology 53:1755–1765.

Sugihara, N. G., J. W. van Wagtendonk, and J. Fites-Kaufman. 2018. Fire as an ecological process. Pages 58–74 *in* J. W. van Wagtendonk, N. G. Sugihara, S. L. Stephens, A. E. Thode, K. E. Shaffer, and J. A. Fites-Kaufman, editors. Fire in California's ecosystems. Second edition. University of California Press, Berkeley, California, USA.

Swanson, M. E., J. F. Franklin, R. L. Beschta, C. M. Crisafulli, D. A. DellaSala, R. L. Hutto, D. B. Lindenmayer, and F. J. Swanson. 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment 9:117–125.

Swetnam, T. W., C. D. Allen, and J. L. Betancourt. 1999. Applied historical ecology: using the past to manage for the future. Ecological Applications 9:1189–1206.

Swetnam, T., and C. Baisan. 1996. Historical fire regime patterns in the southwestern United States since 1700. Pages 11–32 *in* C. D. Allen, editor. Fire Effects in Southwestern Forests, Proceedings of the Second La Mesa Fire Symposium, March 29–31, 1994, Los Alamos, New Mexico. RM-GTR-286. U.S. Department of Agriculture, Rocky Mountain Forest and Range Experiment Station.

Swetnam, T. W., C. H. Baisan, A. C. Caprio, P. M. Brown, R. Touchan, R. S. Anderson, and D. J. Hallett. 2009. Multi-millennial fire history of the Giant Forest, Sequoia National Park, California, USA. Fire Ecology 5:120–150.

Swetnam, T. L., and P. M. Brown. 2010. Comparing selected fire regime condition class (FRCC) and LANDFIRE vegetation model results with tree-ring data. International Journal of Wildland Fire 19:1–13.

Swetnam, T. W., J. Farella, C. I. Roos, M. J. Liebmann, D. A. Falk, and C. D. Allen. 2016. Multiscale perspectives of fire, climate and humans in western North America and the Jemez Mountains, USA. Philosophical Transactions of the Royal Society B 371:20150168.

Taylor, A. H. 2000. Fire regimes and forest changes in mid and upper montane forests of the southern Cascades, Lassen Volcanic National Park, California, U.S.A. Journal of Biogeography 27:87–104.

Taylor, A. H., C. Airey-Lauvaux, B. Estes, L. Harris, and C. N. Skinner. 2020. Spatial patterns of nineteenth century fire severity persist after fire exclusion and a twenty-first century wildfire in a mixed conifer forest landscape, Southern Cascades, USA. Landscape Ecology 35:2777–2790.

Taylor, A. H., and C. N. Skinner. 2003. Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecological Applications 13:704–719.

Taylor, A. H., V. Trouet, C. N. Skinner, and S. Stephens. 2016. Socioecological transitions trigger fire regime shifts and modulate fire–climate interactions in the Sierra Nevada, USA, 1600–2015 CE. Proceedings of the National Academy of Sciences USA 113:13684–13689.

Tepley, A. J., and T. T. Veblen. 2015. Spatiotemporal fire dynamics in mixed-conifer and aspen forests in the San Juan

Mountains of southwestern Colorado, USA. Ecological Monographs 85:583–603.

Thompson, J. R., T. A. Spies, and L. M. Ganio. 2007. Reburn severity in managed and unmanaged vegetation in a large wildfire. Proceedings of the National Academy of Sciences USA 104:10743–10748.

Trant, A., E. Higgs, and B. M. Starzomski. 2020. A century of high elevation ecosystem change in the Canadian Rocky Mountains. Scientific Reports 10:9698.

Turner, M. G. 1989. Landscape ecology: the effect of pattern on process. Annual Review of Ecology and Systematics 20:171–197.

Van Horne, M. L., and P. Z. Fulé. 2006. Comparing methods of reconstructing fire history using fire scars in a southwestern United States ponderosa pine forest. Canadian Journal of Forest Research 36:855–867.

van Mantgem, P. J., D. A. Falk, E. C. Williams, A. J. Das, and N. L. Stephenson. 2020. The influence of pre-fire growth patterns on post-fire tree mortality for common conifers in western US parks. International Journal of Wildland Fire 29:513–518.

van Wagtendonk, J. W., J. A. Fites-Kaufman, H. D. Safford, M. P. North, and B. M. Collins. 2018*a*. Sierra Nevada bioregion. Pages 249–278 *in* J. van Wagtendonk, N. G. Sugihara, S. L. Stephens, A. E. Thode, K. E. Shaffer, and J. Fites-Kaufman, editors. Fire in California's ecosystems. Second edition. University of California Press, Berkeley, California, USA.

van Wagtendonk, J. W., N. G. Sugihara, S. L. Stephens, A. E. Thode, K. E. Shaffer, and J. Fites-Kaufman, editors. 2018*b*. Fire in California's ecosystems. Univ of California Press, Berkeley, California, USA.

Veblen, T. T., and J. A. Donnegan. 2005. Historical range of variability for forest vegetation of the national forests of the Colorado Front Range. Final report. University of Colorado, Boulder, Colorado, USA.

Veblen, T. T., and D. C. Lorenz. 1991. The Colorado front range: a century of ecological change. University of Utah Press, Salt Lake City, Utah USA.

Walker, R. B., J. D. Coop, W. M. Downing, M. A. Krawchuk, S. L. Malone, and G. W. Meigs. 2019. How much forest persists through fire? High-resolution mapping of tree cover to characterize the abundance and spatial pattern of fire refugia across mosaics of burn severity. Forests 10:782.

Walker, R. B., J. D. Coop, S. A. Parks, and L. Trader. 2018*a*. Fire regimes approaching historic norms reduce wildfire-facilitated conversion from forest to non-forest. Ecosphere 9:e02182.

Walker, R. B., J. D. Coop, S. A. Parks, and L. Trader. 2018*b*. Fire regimes approaching historic norms reduce wildfire-facilitated conversion from forest to non-forest. Ecosphere 9:e02182.

Wallin, D. O., F. J. Swanson, B. Marks, J. H. Cissel, and J. Kertis. 1996. Comparison of managed and pre-settlement landscape dynamics in forests of the Pacific Northwest, USA. Forest Ecology and Management 85:291–309.

Warde, W., and J. W. Petranka. 1981. A correction factor table for missing point-center quarter data. Ecology 62:491–494.

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increase Western U.S. forest wildfire activity. Science 313:940–943.

Whitlock, C., and P. J. Bartlein. 1997. Vegetation and climate change in northwest America during the past 125 kyr. Nature 388:57–61.

Whitlock, C., C. N. Skinner, P. J. Bartlein, T. Minckley, and J. A. Mohr. 2004. Comparison of charcoal and tree-ring records of recent fires in the eastern Klamath Mountains, California, USA. Canadian Journal of Forest Research 34:2110–2121.

Wiens, J. A., G. D. Hayward, H. D. Safford, and C. Giffen. 2012. Historical environmental variation in conservation and natural resource management. John Wiley & Sons, West Sussex, UK.

Williams, M. A., and W. L. Baker. 2011. Testing the accuracy of new methods for reconstructing historical structure of forest landscapes using GLO survey data. Ecological Monographs 81:63–88.

Williams, M. A., and W. L. Baker. 2012. Spatially extensive reconstructions show variable-severity fire and heterogeneous structure in historical western United States dry forests. Global Ecology and Biogeography 21:1042–1052.

Wimberly, M. C., and R. S. H. Kennedy. 2008. Spatially explicit modeling of mixed-severity fire regimes and landscape dynamics. Forest Ecology and Management 254: 511–523.

Winthers, E., et al. 2005. Terrestrial ecological unit inventory technical guide. General Technical Report WO-GTR-68. U.S. Department of Agriculture, Forest Service, Washington Office, Ecosystem Management Coordination Staff, Washington, D.C., USA.

Wisdom, M. J. 2000. Source habitats for terrestrial vertebrates of focus in the Interior Columbia Basin: Broad-scale trends and management implications. PNW-GTR-485. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Wu, J., and O. L. Loucks. 1995. From balance of nature to hierarchical patch dynamics: a paradigm shift in ecology. Quarterly Review of Biology 70:439–466.

Yocom Kent, L. 2014. An evaluation of fire regime reconstruction methods. Working Papers in Southwestern Ponderosa Pine Forest Restoration 32. Northern Arizona University, Flagstaff, Arizona, USA. http://www.nwfirescience.org/sites/default/files/publications/doc.pdf

Yocom Kent, L. L., and P. Z. Fulé. 2015. Do rules of thumb measure up? Characteristics of fire-scarred trees and samples. Tree-Ring Research 71:78–82.

Yocom Kent, L. L., et al. 2017. Climate drives fire synchrony but local factors control fire regime change in northern Mexico. Ecosphere 8:e01709.

Yocom-Kent, L. L., P. Z. Fulé, W. A. Bunn, and E. G. Gdula. 2015. Historical high-severity fire patches in mixed-conifer forests. Canadian Journal of Forest Research 45:1587–1596.

Young, D. J. N., J. T. Stevens, J. M. Earles, J. Moore, A. Ellis, A. L. Jirka, and A. M. Latimer. 2017. Long-term climate and competition explain forest mortality patterns under extreme drought. Ecology Letters 20:78–86.

Supporting Information

Additional supporting information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/eap.2431/full

Ecological Applications, 31(8), 2021, e02432

Wildfire and climate change adaptation of western North American forests: a case for intentional management

PAUL F. HESSBURG,[1,2,6] SUSAN J. PRICHARD,[2] R. KEALA HAGMANN,[2,3] NICHOLAS A. POVAK,[1,4] AND FRANK K. LAKE[5]

[1] *USDA-FS, Pacific Northwest Research Station, 1133 N. Western Avenue, Wenatchee, Washington 98801 USA*
[2] *College of the Environment-SEFS, University of Washington, Seattle, Washington 98195 USA*
[3] *Applegate Forestry LLC, Corvallis, Oregon 97330 USA*
[4] *USDA-FS, Pacific Southwest Research Station, 2480 Carson Road, Placerville, California 95667 USA*
[5] *USDA-FS, Pacific Southwest Research Station, 1700 Bayview Drive, Arcata, California 95521 USA*

Citation: Hessburg, P. F., S. J. Prichard, R. K. Hagmann, N. A. Povak, and F. K. Lake. 2021. Wildfire and climate change adaptation of western North American forests: a case for intentional management. Ecological Applications 31(8):e02432. 10.1002/eap.2432

Abstract. Forest landscapes across western North America (wNA) have experienced extensive changes over the last two centuries, while climatic warming has become a global reality over the last four decades. Resulting interactions between historical increases in forested area and density and recent rapid warming, increasing insect mortality, and wildfire burned areas, are now leading to substantial abrupt landscape alterations. These outcomes are forcing forest planners and managers to identify strategies that can modify future outcomes that are ecologically and/or socially undesirable. Past forest management, including widespread harvest of fire- and climate-tolerant large old trees and old forests, fire exclusion (both Indigenous and lightning ignitions), and highly effective wildfire suppression have contributed to the current state of wNA forests. These practices were successful at meeting short-term demands, but they match poorly to modern realities. Hagmann et al. review a century of observations and multi-scale, multi-proxy, research evidence that details widespread changes in forested landscapes and wildfire regimes since the influx of European colonists. Over the preceding 10 millennia, large areas of wNA were already settled and proactively managed with intentional burning by Indigenous tribes. Prichard et al. then review the research on management practices historically applied by Indigenous tribes and currently applied by some managers to intentionally manage forests for resilient conditions. They address 10 questions surrounding the application and relevance of these management practices. Here, we highlight the main findings of both papers and offer recommendations for management. We discuss progress paralysis that often occurs with strict adherence to the *precautionary principle*; offer insights for dealing with the common problem of irreducible uncertainty and suggestions for reframing management and policy direction; and identify key knowledge gaps and research needs.

Key words: Climate Change and Western Wildfires; climate warming; forest landscape changes; Indigenous fire use; landscape realignment; landscape resilience; landscape resistance; social-ecological systems; wildfire regime changes.

INTRODUCTION

Western forests are rapidly changing

Starting in the mid-1980s, area burned in seasonally dry forests of western North America (wNA) began a steady rise (Westerling et al. 2006), despite increasing investment in fire suppression (Calkin et al. 2005). Seasonally dry forests are those pine, dry or moist mixed-conifer, and cold forests that are available to burn most years during the wildfire season (refer to forest type definitions and discussion in Hessburg et al. 2019). Increased burned area is attributed to combinations of warmer seasonal temperatures, longer wildfire seasons, drier summers, below-average winter precipitation and earlier snowmelt, and increasing human ignitions (Westerling et al. 2006, Morgan et al. 2008). The incidence of large wildfires has likewise increased across wNA over the last three decades (Schoennagel et al. 2017, Parks and Abatzoglou 2020), while burned area in the Inland Northwest and American Southwest has risen most noticeably over the last two decades (Westerling 2016). These increases are occurring not only in dry pine and mixed-conifer forests, but also in moist and cold

Manuscript received 18 November 2020; revised 17 February 2021; accepted 3 March 2021; final version received 6 July 2021. Corresponding Editor: David S. Schimel.
[6] E-mail: paul.hessburg@usda.gov

forests, and in nearby nonforest vegetation (preforest, grassland, shrubland, and sparse woodland; Parks et al. 2015). Based on climate change predictions, burned area in wNA will at least double or triple by mid-century (McKenzie et al. 2004, Westerling et al. 2011).

While increase in burned area is strongly associated with climatic warming, changes to other aspects of wildfire regimes (Table 1) more directly reflect the influence of human activities. For example, in many wNA forests, land and resource management decisions and actions led to abrupt and persistent declines in fire frequency (and hence, burned area) beginning more than 170 yr ago. Decreased fire frequency led to increased continuity and accumulation of live and dead fuels (Stephens et al. 2009), both of which contribute to increases in fire severity as burned area increases (Parks and Abatzoglou 2020). Likewise, while human-caused ignitions continue to contribute to increasing burned area (Balch et al. 2017), they also reflect contemporary land development and access patterns. With ongoing fire suppression, lengthening wildfire seasons, and the increased likelihood of extreme fire weather, fire effects are broadly becoming more severe than those experienced in the last two centuries (North et al. 2015, Parks and Abatzoglou 2020). As a result, forests developing after large contemporaneous wildfires little resemble forests evolving under a more characteristic wildfire regime (Keane et al. 2002, Hessburg et al. 2005, Coop et al. 2020).

The challenge of larger and more intense wildfires

The increasing impacts that large and intense wildfires will have on social and ecological systems will be the major challenge facing managers of seasonally dry forests over the 21st-century. Prolonged smoke production and human health effects, chronic soil erosion and mass wasting, degraded water supplies, loss of cultural and natural resources, increased greenhouse gas emissions and reduced carbon storage are all growing issues (Spies et al. 2014). Management capacity to influence how much area burns will be somewhat limited (cf. Taylor et al. 2016), but fuel reduction treatments, including prescribed burning, coupled forest thinning and prescribed burning, and managed wildfires, are proven methods to influence the ecological impacts of wildfire, and mitigate impacts of extreme fire events on social systems (Taylor et al. 2016; Prichard et al. 2021). To date, mechanical fuel reduction treatments have been applied to small portions of wNA forested landscapes (Barnett et al. 2016, Vaillant and Reinhardt 2017, Kolden 2019, Kolden and Henson 2019). One reason is that land allocations amenable to mechanical treatments (via their enabling legislation) represent a dwindling fraction of public lands (Fig. 1); another is a lack of sufficient experience with prescribed burning and managing wildfires in front or backcountry locations. However, scaling-up a broad variety of fuel reduction treatments can tip landscape dynamics in favor of more benign fire behavior and effects (Stevens et al. 2014, Parks et al. 2016, Taylor et al. 2016, Ager et al. 2020).

Table 1. Components of an active fire regime.

Component	Definition
Amount	total amount of area burned annually or decadally
Distribution (severity)	distribution of severity class patch sizes
Distribution (event areas)	distribution of fire event patch sizes
Frequency	average fire return interval, and variation around the mean
Spatial distribution	the geographic distribution of fires†
Intensity	the energy release from surface and crown fires at the flaming front
Duration	the length of time fires burn‡
Seasonality	the time of the year when fires burn

Note: Components may vary by climatic period.
†The spatial distribution of fires is dictated by biophysical setting, climate, and weather conditions, forest or nonforest type, ignition probability, and the propensity for reburning.
‡The period of fires is dependent on the climate, weather, and fuel bed characteristics.

The assertion of regional-scale adaptation needs

The need for broad-scale climate and wildfire adaptation across wNA is predicated on two main assertions. The first is that most seasonally dry forest landscapes, and some drier coastal forests (Hessburg et al. 2019), have significantly changed over at least the last two centuries under the influences of curtailed Indigenous burning before 1850 (Kay 2000, Stewart 2002); wildfire exclusion (beginning with domestic livestock grazing in the mid-1850s, Belsky and Blumenthal 1997); and decades of selection cutting of large, old, early seral tree species (Hessburg and Agee 2003, Lydersen et al. 2013). The resultant stand- to landscape-scale changes in forest structures and fuels have left these seasonally dry forests vulnerable to the direct and indirect effects of climate warming, drought, and wildfire (Allen et al. 2002, Noss et al. 2006, Keane et al. 2018, Bryant et al. 2019, Hessburg et al. 2019). The second assertion is that climate change and wildfire adaptation treatments implemented at large regional landscape scales can effectively moderate many ecosystem transitions, conserve greater area and heterogeneity of forest successional conditions (Moritz et al. 2013, Coop et al. 2020), better foster native biodiversity (Raphael et al. 2001, Bisson et al. 2003, Isaak et al. 2010, Rieman et al. 2010), and maintain essential and desirable ecosystem services and processes (e.g., see Dale et al. 2001, Millar et al. 2007, Hurteau et al. 2014).

Public land management: political and paralyzed

As with many topics in conservation biology (Soulé 1985), active or intentional management of public

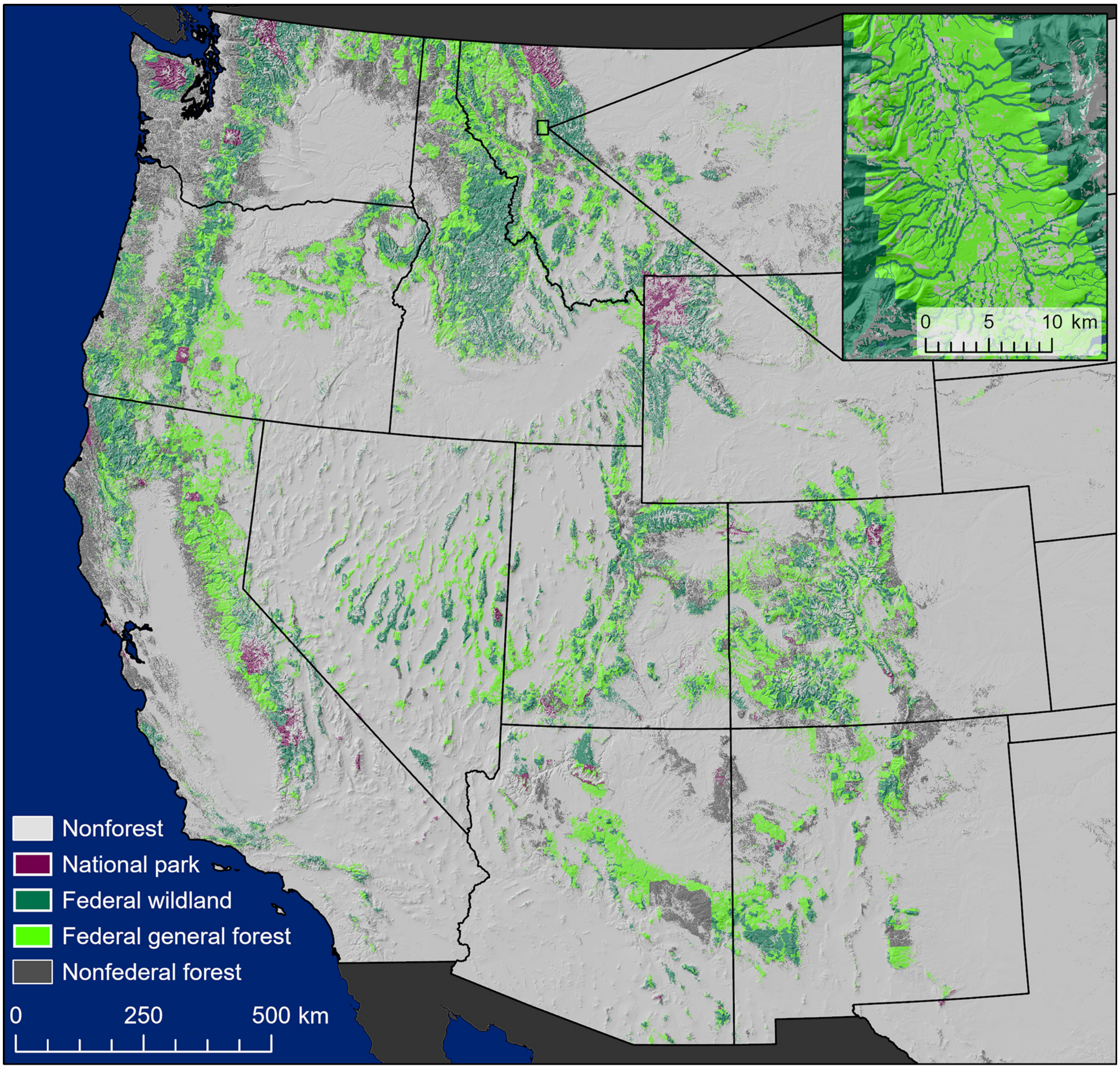


Fig. 1. Map of forested areas and their primary public land management allocations in the western United States. Federal wildlands include administratively withdrawn roadless areas, congressionally designated wilderness, and terrestrial habitat reserve networks. General forest areas are those remaining that are ostensibly amenable to mechanical thinning and prescribed burning treatments. Riparian reserves are generally not shown due to map scale, but they represent a significant area in general forest. The inset map at top right shows an example of riparian reserves in the Swan sub-basin of northwest Montana. Riparian buffers are 100 m on either side of perennial streams and 30 m on ephemeral streams. Most federal wildlands and national parks are available in concept for using managed wildfires and prescribed burning as fuel reduction treatments, but application of these tools remains uneven. Data sources for map development are (1) for forested areas, National Land Cover Database, NLCD (2006); https://www.mrlc.gov/data/nlcd-2006-land-cover-conus) for inventoried Roadless Areas (2001), https://www.fs.usda.gov/detail/roadless/2001roadlessrule/maps/?cid=stelprdb5382437, USDA-FS internal enterprise data layer name: S_USA. InvRoadlessArea_2001; (2) for Northwest Forest Plan Land Use Allocations (2013), https://www.fs.fed.us/r6/reo/landuse/, USDA-FS internal enterprise data layer name: S_R06.NWFP_LandUseAllocation_2013; (3) for designated Wilderness Areas (2020), USDA-FS internal enterprise data layer name: S_USA.Wilderness; (4) for Other National Designated Areas (2020), USDA-FS internal enterprise data layer name: S_USA.PADUS_DESIGNATION; (5) for US National Atlas Federal and Indian Land Areas (last updated 2004), USDA-FS internal enterprise data layer name: S_USA.OtherNationalDesignatedArea.

forestlands often devolves into value-laden discussions and politicized views of appropriate contexts and frames of reference (Peery et al. 2019). Here, we define intentional management as the planned application of silvicultural and prescribed fire treatments and managed wildfire to meet a variety of specific landscape-level objectives in predefined conditions and contexts. This can include opportunities in Indigenous communities for more decentralized stewardship practices related to resource tending, subsistence activities, and spiritual or

religious observances (Norgaard 2014). In contrast, under passive management (i.e., continued fire suppression with little intentional forest or fuel management), many assume that existing conditions and processes will naturally sort out an effective remedy without benefit of intentional management (Carey 2006).

Owing to extensive 20th-century harvest of large trees and old-growth forests, public trust in forest management eroded. In response, an emphasis emerged to conserve the victims of that history, large trees and old forests, and native species and their habitats, by minimizing further forest management actions. This emphasis is commonly underscored with policies that emphasize riparian and terrestrial habitat reserves and related conservation areas (Spies et al. 2019, Stephens et al. 2019), by legal injunction of active management projects (Prichard et al. 2021), and by maintaining a relatively small footprint of areas available for active management (Fig. 1). Opposition to active forest management is reflected in statements like "active forest management is unwarranted because the effects of fire exclusion and forest changes are overstated; . . .is ineffective and counterproductive; . . . should be focused on the wildland urban interface; or wildfires alone can do the work of fuel treatments." On the other hand, support for active management from the commercial sector suggests that "forest thinning alone can mitigate wildfire severity; forest thinning and prescribed burning can solve the problem; or managed wildfires hold no real promise." Each statement polarizes debates and oversimplifies the problems and the solutions (Prichard et al. 2021). Given rapid climate change and a legacy of excluding most natural and Indigenous ignitions, effective forest landscape restoration and adaptation strategies are more complex and nuanced than any of these statements imply.

Advocacy and objectivity: is it one or the other?

The polarization and politicization of scientific evidence impedes implementation of effective land management plans, policies, and management by raising the volume of the disagreement; obscuring the line between facts, opinions, and legal requirements; creating the impression that knowledge is more uncertain than it is; and increasing the time to resolution. Wellerstein (2018) argue that the premise of science as apolitical is simply a myth, since all science takes place and is supported within a highly political environment. Nonetheless, when scientists affiliate themselves with one-sided or partisan views and activism, they inevitably minimize their value and that of the applied science (Lackey 2007, Pielke 2007).

Scientists are increasingly asked to comment on forest policy and management recommendations. Facilitating communication among stakeholders of public land management by providing practical access to the best-available science can more effectively ensure scientifically credible decision-making (Komatsu and Kume 2020). However, while some encourage scientists to be responsible informants for species or ecosystem conservation (e.g., Lach et al. 2003), others worry that their objectivity in conservation or ecological research may be compromised (e.g., Scott et al. 2007), especially during volatile times, and with arguments that are already polarized or politicized.

Garrard et al. (2016) argue that scientists are not compromised when they transparently evaluate policies or recommendations for their consistency with the best available science, its weight of evidence, and any associated uncertainties. A systematic evaluation of best available science would include careful examination of Indigenous and western data, information, knowledge, and wisdom from a variety of locally and regionally relevant sources (Varghese and Crawford 2020). Garrard et al. (2016) further suggest that in the face of serious societal, economic, or existential issues, "the standard of debate about conservation is impoverished when scientists with relevant knowledge remain silent outside the pages of their academic journals."

Peery et al. (2019) provide a framework for evaluating agenda-driven science and a case example of controversy in the scientific literature that has impacted management of the California spotted owl and its habitats. They discuss professional norms for scientist engagement with management and policy issues and conclude "that intentionally engaging in activities outside of these professional norms to promote desired political outcomes, as part of either the production or dissemination of science . . .constitute[s] agenda-driven science."

Recent controversy involving the creation and dissemination of agenda-driven science is creating uncertainty for forest managers and policy-makers throughout wNA. Contributing to the controversy are publications that challenge the significance of forest condition and wildfire regime changes, and the advisability of proactive management without addressing the core arguments (e.g., compare Hessburg et al. [2020] and Mildrexler et al. [2020] and their discussion of trade-offs between wildfire dynamics, carbon sequestration, and forest adaptation to climate warming). To aid those engaged in designing, evaluating, and implementing science-based adaptation options, we examine the strength of evidence pertaining to these topics.

We first provide a framework for characterizing and evaluating changes in forest conditions and fire regimes in Hagmann et al. (2021). We then review the strength of evidence documenting changes or lack thereof. Similarly, in Prichard et al. (2021), we review the strength of evidence surrounding the usefulness of various passive and active management treatments to provide remedies to current conditions. We then discuss 10 key questions related to application of methods as viable treatments.

Forest Conditions and Wildfire Regimes

Advances in fire and landscape science over the past several decades enable rigorous multi-proxy and multi-

scale assessments of variation in historical fire regime and forest vegetation conditions. These insights build on more than a century of assessment of changing fire behavior and forest landscape conditions. Beginning in the 1930s, fire histories based on tree-ring and fire-scar records have provided high-resolution, cross-dated, multi-centenary evidence of the spatial point patterns of fires, which have enabled precise interpretations of fire frequency associated with recorder trees. While fire-scar records remain a primary means of exploring historical fire ecology, more recently developed methodologies and multi-proxy assessments have expanded the potential to evaluate broader temporal and spatial patterns. New insights into complex relations between variations in climate, fire, and vegetation emerge from multi-proxy and trans-disciplinary studies that combine sedimentary pollen and charcoal records (Higuera et al. 2007), large-scale tree cohort analyses (Schoennagel et al. 2011), early 20th-century land system inventories and surveys (Hagmann et al. 2013, 2014, Levine et al. 2017), landscape reconstructions from historical black and white photographic imagery (oblique, panoramic, stereo photo pairs, Hessburg et al. 1999, 2000, Stockdale et al. 2015, 2019), forest inventories and land system surveys (Williams and Baker 2012, cf. Fulé et al. 2014, Odion et al. 2014, cf. Stevens et al. 2016), and simulation modeling of landscape succession and disturbance regimes (Keane et al. 2004, 2018, McGarigal and Romme 2012). Additionally, trans-disciplinary studies that employ fire-scar research, climate, archaeological, and ethnographic studies show that many different Indigenous cultures were significant contributors to the magnitude and extent of fire influence on the wNA landscape (e.g., see Taylor et al. 2016, Lightfoot et al. 2013).

Over the past two decades, a series of publications using novel techniques has suggested that 19th- to 21st-century changes in western forests and their fire regimes have been less substantial than a much larger and more diverse body of scientific evidence has long indicated. Hagmann et al. (2021) provide a comprehensive review of these papers and studies that directly evaluated them. They show that methods and inferences in these articles failed independent validation by other research groups and lend their support to the findings of the larger body of evidence.

The evidence for change in forest conditions

Hagmann et al. (2021) relied on several hundred research articles from research groups throughout wNA that examined historical changes to seasonally dry forests patterns and processes to illustrate key departures from conditions that existed prior to European colonization. They found that changes in forest successional landscapes are significant in all forest types, whether dry, moist, or cold. Changes are prominent at tree, patch, and local and regional landscape levels, and these changes explain important shifts in numerous habitats and ecosystem processes. Conditions of nonforest vegetation (grasslands, shrublands, sparse woodlands) are likewise altered as a consequence of fire exclusion and forest encroachment. While some forest and nonforest ecosystems may not have been directly altered by fire exclusion, the magnitude of changes suggests that it is likely that all were indirectly impacted by alteration of the landscape ecology and disturbance regimes that surround them. Based on a preponderance of scientific evidence, there can be little doubt that long-term fire exclusion and other associated social-ecological influences contributed to extensive modification of landscapes across wNA, and that the magnitude of the departures in fire regimes and landscape conditions has increased the vulnerability of contemporary forested landscapes to fire and drought-related stressors.

The evidence for change in fire regimes

Hagmann et al. (2021) also review the evidence for changes in the dimensions of fire regimes (Table 1). Fire exclusion has reduced fire frequency in all forest types, with the degree of change generally declining with increasing elevation, owing to orographic effects on moisture and temperature, and topo-edaphic effects on insolation. As a consequence, surface and ladder fuel abundance generally increased in historically fuel-limited frequent-fire forests, while forest cover at higher elevations expanded and became more successionally homogenized (Fig. 2). In both cases, crownfire vulnerability increased. Long-term fire exclusion reduced the total amount and spatial distribution of wildfires resulting in a nearly universal fire deficit in forests (Parks et al. 2015, Parks and Abatzoglou 2020).

Adapting Forests to Wildfires and Climate Change

Prichard et al. (2021) address 10 key questions surrounding active forest management, address the assumption that historical fire regimes were "natural" rather than cultural, and describe conditions where specific management actions are appropriate and effective for adapting current forests to wildfires and climate change. The authors again use a strength of scientific evidence approach to show why the answers to the 10 questions are relatively straightforward. In addition to evaluating the efficacy of diverse treatments to moderate expected fire severity, they discuss these questions in the context of their consistency with more holistic climate- and wildfire-adaptation strategies that are designed to achieve many social and ecological benefits. Moreover, they discuss how methods designed to achieve a single objective often fail given contemporary goals for multi-objective landscape management. We summarize their responses to the 10 questions here.

1) *Are the effects of fire exclusion overstated? If so, are treatments unwarranted and even counterproductive?*

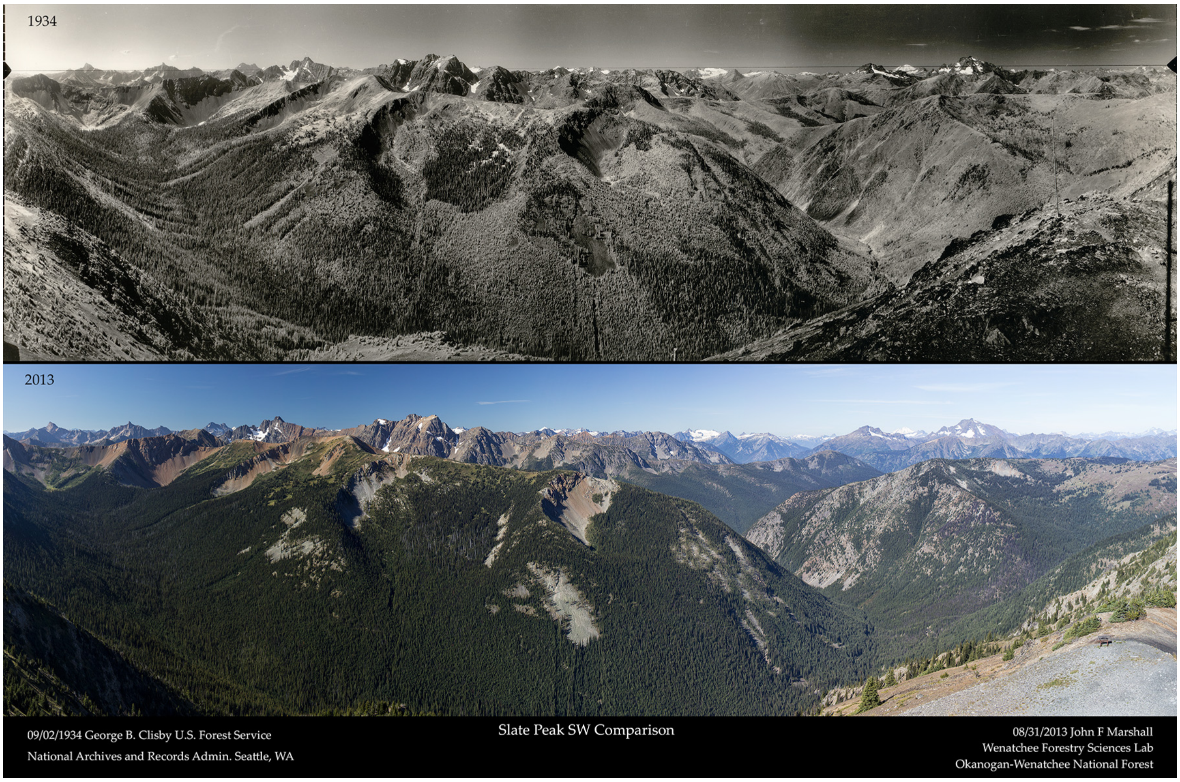


FIG. 2. Top photo: View from atop Slate Peak in northeastern Washington, looking southwest, 1934, George Clisby photograph, National Archives, Seattle, Washington, USA. The 1934 panoramic view shows extensive evidence of prior wildfires, varied age classes of cold forest, and recently burned and recovering areas. In the same view nearly eight decades later (bottom photo, 2013, John Marshall Photography), note the complete absence of recent fire evidence, widespread ingrowth creating denser forests, loss of nonforest, and lack of forest successional heterogeneity.

Prichard et al. (2021) dispute all parts of this question. They reveal four crucial components in their answer, not the least of which is that increasing forest resilience and resistance to wildfires and climate change provides positive societal and ecosystem benefits, which overwhelm uncertainties about prior historical conditions. They also show that intentional forest management is effective and corrective where practiced, but its pace and footprint are insufficient to current needs.

2) *Is forest thinning alone sufficient to mitigate wildfire hazard?* Whether forest thinning achieves adaptation objectives depends on several factors, including its timing, location, rate, and spatial scale. Reducing canopy layering and density limits crownfire potential, but unless the abundance and connectivity of surface fuels and fuel ladders is also reduced, thinning can have limited effectiveness in mitigating wildfire severity, and may make matters worse. In forest thinning for adaptation to climate change and wildfires, emphasis is placed on residual forest structure, composition, and understory fuels rather than on the trees that are removed (Larson and Churchill 2012, Churchill et al. 2013).

3) *Can forest thinning and prescribed burning achieve climate adaptation?* Coupled thinning and prescribed burning treatments are proven approaches to mitigating wildfire severity in many seasonally dry forests, but they are not appropriate to all forest types, land allocations, and conditions. These treatments require regular maintenance application of prescribed or cultural burning to maintain low surface fuel levels and remove developing fuel ladders. The vast scale of ongoing fuel reduction necessitates wise use of naturally ignited future fires during moderate fire weather as well.

4) *Should active forest management, including forest thinning, be concentrated in the wildland urban interface (WUI)?* Fuel treatments in the WUI are critically important as is reducing continued development in high fire danger areas (Balch et al. 2017, Radeloff et al. 2018). People living in the WUI have ample incentive to reduce their vulnerability to wildfires (Cohen 2000), and many resources are available to

help them do so (Syphard et al. 2012; *resources available online*).[7,8] However, this logic, concentrating treatments in WUI, fails to address interconnectedness between social and ecological systems in landscapes beyond the WUI. Examples include wildfire emissions and broad-scale smoke movement in the atmosphere; water quality and quantity provided by municipal watersheds distant from population centers; ember attack on WUI areas from wildfires burning several kilometers away; wildfire effects on power, WIFI, and broadband transmission and distribution networks; and tribal connections to ancestral territories and resources. It also avoids core decisions related to human social values for forested landscapes and long-term ecosystem dynamics. Alternatively, intentional forest management both within and beyond the WUI offers the greatest social and ecological benefits.

5) *Can wildfires on their own do the work of fuel treatments?* Many forests are experiencing rapid WUI expansion, leaving land managers and citizens increasingly unwilling to accept the risks of managed wildfires In the backcountry, some wildfires are allowed to burn under specified conditions to achieve incident and resource management objectives. However, the effects of fire exclusion are varied and extensive, and managed wildfire is a spatially imprecise tool. To increase predictability of outcomes, application during benign to moderate fire weather may be preferred; this, however, necessitates numerous follow-up treatments to meet objectives, and it broadens the period of landscape vulnerability to more extreme wildfires. Considering the narrow seasonal operating window and spatial imprecision concerns, managed wildfires cannot be a cure-all, but can be one of several options in a broader toolkit.

6) *Is the primary objective of fuel reduction treatments to assist in future firefighting response and containment?* The central objective of fuel treatments is to moderate fire behavior when fire inevitably returns, not to stop fire spread or reduce ignitions If fuel treatments simply improve suppression success, less area is burned in the short term but more area will escape control in the future, resulting in deferred risk and contributing to larger and more often severe wildfires.

7) *Do fuel treatments work under extreme fire weather?* Many studies show that fuel reduction treatments are effective at moderating subsequent fire severity, even under extreme weather Far fewer experimental or empirical studies challenge this premise. Moreover, there is strong evidence that some prior burn and reburn mosaics reduce landscape contagion, which limits subsequent spread and severity of wildfires.

[7] https://www.nfpa.org/Public-Education/Fire-causes-and-risks/Wildfire/Preparing-homes-for-wildfire.

[8] https://www.usfa.fema.gov/wui_toolkit/wui_planning.html.

8) *Is the scale of the problem too great? Can we ever catch up?* Given the scale of the area burned by wildfires each decade compared to area treated, some surmise that fuel treatments are futile Nevertheless, a large body of work shows that fuel reduction treatments, including portions of some past wildfires, effectively mitigate subsequent fire behavior and effects. It follows that strategies can be developed to increase rather than decrease the role of fuel reduction treatments. The key to defining the locations, spatial scale, timing, rate of treatment, and methods used is the desired forest–nonforest conditions that result, their degree of adaptation to changing climate and fire regime conditions, and the degree of comfort with spatial uncertainty of outcomes. Moreover, coupling Indigenous cultural burning, fuel reduction, and prescribed fire treatments furthers Indigenous fire stewardship and food security (Sowerwine et al. 2019*a*,*b*), and recovers opportunities for tribal engagement in resource management within their ancestral territories (Long and Lake 2018, Long et al. 2020).

9) *Will planting more trees in wNA forests help to mitigate climate change?* Widespread tree planting has been proposed as a key climate change mitigation (Bastin et al 2019). This premise has poor scientific support in many fire-prone regions of the world (e.g., Grainger et al. 2019, Lewis et al. 2019, Veldman et al. 2019). An increasing body of evidence reveals that proactive management to restore more resilient forest and nonforest structure and composition over large areas, and diversifying tree planting species mixes, can more effectively maintain or increase carbon stores relative to the effects of modern wildfires (e.g., see Hof et al. 2017).

10) *Is post-fire management needed or even ecologically justified?* Prior to fire exclusion, historical landscapes in seasonally dry regions of wNA were the product of complex mosaics of low-, moderate- and high-severity fire patches, which yielded highly variable patterns of surviving forest and scattered fire refugia (ie., unburned patches that functioned as seed sources for postfire tree regeneration in their vicinity). After contemporaneous wildfires, this mosaic is often simplified by large high-severity fire patches, and fire refugia are operationally burned out in closing suppression actions. Within one to two decades after a high-severity fire, dead wood accumulations contribute to uncharacteristically high surface fuel loads. Post-fire removal of the dead understory stems (i.e., those that had previously colonized the landscape during the lengthy period of fire exclusion) by harvest or reburning can mimic this historical reburn influence, thereby minimizing surface fuels in some developing new forests (Stevens-Rumann and Morgan 2016), and reducing future wildfire vulnerability (Coppoletta et al. 2016). The ecological justification for this post-fire removal of the smaller dead understory trees can be

observed in the low surface fuel loads associated with the frequent reburning of pre-management era landscapes and modern-day wilderness areas. It is also clearly revealed in intentional Indigenous cultural burning practices. Indigenous fire stewardship actively mediated post-fire landscape effects to stagger the availability of desired resources and species over time, and ensure their quality, quantity, and abundance (Boyd 1999, Lake and Christianson 2019).

Strategies for Adapting Western Landscapes

Changes in forested landscapes throughout wNA are somewhat unique geographically, as are the stories of change. To consider appropriate climate and wildfire adaptation strategies, managers are compelled to evaluate current vegetation and fuel conditions, the influences that precipitated changes in conditions, the magnitude of the changes, ecological and social constraints to adaptation, patch to landscape vulnerability to changing climatic and wildfire regimes, and nonnative species and any sensitive or endangered species concerns.

Stephens et al. (2010) recommend four strategies for adapting western landscapes to changing climatic and wildfire regimes, and they can be applied in a variety of contexts. They define *resistance* work as that which mitigates expected wildfire effects and protects valued resources, while *realignment* work modifies existing conditions to restore key ecosystem patterns and the processes they drive. Creating *resilient* conditions improves the natural capacity of an ecosystem to respond favorably when unplanned disturbances occur. Finally, they present *response* work as any intentional facilitation to achieve socially and ecologically desirable results that are otherwise difficult to achieve. Each of these strategies can play a role in proactive management going forward. Where their application also considers Indigenous cultural adaptations to climate, vital ecosystem processes, and active cultural use of fire, there will be greater likelihood that resulting vegetation conditions are strongly linked to culturally valued resources and services (Power et al. 2018).

Whether reactive or proactive fire management

Modern wildfire suppression extinguishes essentially all fire starts except those that overwhelm fire suppression capacity and can only be extinguished when aided by a significant change in the weather (North et al. 2015). Fires burning under extreme fire weather often burn vast areas, much larger than the current footprint of managed wildfires and other fuel treatment projects. As a consequence, wildfires that escape initial suppression efforts burn under the most extreme fire weather conditions and do most of the vegetation management in wNA ecosystems (Calkin et al. 2005, North et al. 2015). Appropriately designed thinning, burning, and managed wildfire treatments, that are tailored to topo-edaphic conditions would be helpful additions to this scenario (sensu Taylor and Skinner 2003, Hessburg et al. 2015). Such treatments would prepare landscapes for wildfires that will inevitably follow.

Managing wildfires that burn under extreme fire weather is a blunt management response, which most often results in failure to meet resource management or conservation objectives. Science-based strategies for forest and fuel management are well known, but lack of social license and sufficient financial and personnel resources currently limit fuel reduction programs to a small percentage of wNA forestlands (Hessburg et al. 2020). Increasing costs of fire suppression and lack of control during large fire growth days reveals a reactive management posture that is progressively prone to failure, despite ever-increasing investments (North et al. 2015, Stephens et al. 2020). Thus, a business-as-usual approach to wildfire in fire-prone regions will not solve the current wildfire dilemma (Moreira et al. 2020). Strategic management of regional landscapes is needed that establishes topographically sensible (sensu Povak et al. 2018 and Taylor and Skinner 2003), fire-maintained, control and anchor points (e.g., see Wei et al. 2019). This would improve fire manager usage of future wildfires as adaptation tools.

A more proactive and evidence-based management goal is to restore active wildland fire regimes and landscape resilience to climate change, and actively enable future wildland fires, prescribed and cultural burning, and managed wildfire to provide a higher standard of social-ecological work. To achieve this goal, massive progress and investment are needed (Madeira and Gartner 2018) to transition management from a reactive to a proactive, forward-looking stance. Such an approach allows for the direct adaptation of wildfire regimes by intentionally crafting landscape patterns that drive more benign fire behavior and less severe drought effects. This will require radically increasing the areal extent of restorative and adaptive fuel reduction treatments as is appropriate to conditions and land allocations. It will also require increased use of natural wildfire ignitions (as above) under moderate fire weather conditions, to recapture the once extensive moderate influences of wildfires, and then maintain that progress with controlled (prescribed and cultural) burning and thinning as needed.

The nested character of regional landscapes holds adaptation clues

The hierarchical organization of historical wNA landscapes influences countless ecosystem functions, including scale-dependent spatial and temporal controls that drive wildfire behavior and effects, and the cross-connection between levels of organization. Characteristics of this organization and its influence on ecosystem

functions can inform realignment of current systems with early 21st-century and projected future climates (Hessburg et al. 2016, 2019). Three important ideas associated with that nested structure are that (1) at a fine spatial scale, species traits and adaptations drive within-patch structure, composition, and response to disturbances; (2) cross-connections between fine-scale patch structure and composition and meso-scale landscape patchworks influence fire frequency and severity because they form the percolation surface where disturbance propagates, and the manner of propagation; and (3) cross-connections between non-forest and forested landscapes mediate broad spatial patterns of fire behavior attributes and their effects. These three ideas help shape a scientifically supported landscape adaptation framework (Hessburg et al. 2015, 2019).

Additionally, we are learning through integration of western science with traditional ecological knowledge that Indigenous fire use and broader landscape stewardship practices were upheld in tribal communities as human services for ecosystems. Indigenous tribes acknowledged and promoted multi-generational contributions to foster landscape resistance and resilience. Trans-generational fire use also promoted post-fire recovery of landscapes and habitats where culturally valued drought-tolerant, fire-adapted plant species were adversely influenced by a fire (Huffman 2013).

Beyond the Precautionary Principle

The precautionary principle holds that when the potential for adverse effects is unknown or difficult to quantify; the burden of proof rests on the proponent of an activity to demonstrate that lack of harm is the most likely outcome. However, it is virtually impossible to demonstrate lack of harm for most any activity, including no action, especially in a rapidly changing environment. Moreover, one-sided, or single-issue application of the principle can overlook desired ecosystem services, species, and processes that proactive work could have protected. Given human influence on climate and wildfire regimes the world over, a no-action alternative that purports to let so-called natural processes like modern wildfires operate unfettered is grossly misleading. These processes are operating within a human-influenced template globally, and their regime characteristics and fuel conditions have been altered by humans, increasing the likelihood that large portions of many modern wildfires are unnatural.

Modern wildfire management dominated by fire suppression is perceived by many as a no-action alternative when compared to active restoration and adaptation in planning and management. However, active suppression of 98% of wildfire ignitions (North et al. 2015) hardly qualifies as no action, as we have shown earlier. The small proportion of wildfires that escape containment all too often rapidly and indiscriminately convert forest to non-forest conditions. This is an altogether unevaluated planning outcome, and the recovery of forest structure and processes can take decades to centuries, if it occurs at all. Furthermore, fire suppression costs currently exceed US$2 billion annually, not including loss of life and property, and detrimental impacts to lifeways, human health, and livelihoods, while the total annualized economic burden of wildfires ranges from US$71 billion to US$347 billion (Thomas et al. 2017; data *available online*).[9]

The precautionary principle is indeed useful guidance, but it must be applied equally to what are often mistakenly perceived as no-action alternatives. Lacking this clarity, broad application of the precautionary principle as a conservation approach can result in greater long-term harm than more ecologically intuitive remedies, as can be seen within the Northwest Forest Plan area of the eastern Cascade and Klamath Mountain regions (Spies et al. 2019, Stephens et al. 2019). There, networks of late-successional reserves (LSRs) for the northern spotted owl in seasonally dry, historically frequent-fire forests increase the likelihood of their elimination by extreme wildfire events (Henson et al 2013, Spies et al. 2018, 2019). There is simply too much at stake to require unattainable certainty about potential risk of harm or losses (Wood and Jones 2019).

The precautionary principle can become the "paralyzing principle" given irreducible uncertainty about risk of loss associated with action and no-action alternatives (Sunstein 2003). The loss of ˜30 million mature and old pine trees during a recent extreme drought in south-central California (Asner et al. 2016) is a stark reminder of the pitfall of requiring unduly high certainty despite decades of established science showing the efficacy of treatments that foster resilient forest structure and composition (Henson et al. 2018, Fettig et al. 2019). Absent 150–170 yr of frequent fires, overgrown forest density conditions produced a massive and predictable die-off event, facilitated by tree-killing bark beetles and drought, that proactive implementation of climate- and wildfire-adaptation strategies could have mitigated (Stephens et al. 2018, Fettig et al. 2019). Remedying such conditions would have required careful consideration of changes over the period of fire exclusion, the effects of climatic changes looking forward, and any related ESA (Endangered Species Act) concerns.

Dealing with uncertainty

There is much uncertainty to science, including that surrounding our knowledge of historical and contemporary forest ecology, future conditions, and adaptive forest management. In that light, active forest management projects with objectives of restoring more resilient and resistant structure and composition can be assessed using a set of questions to address the relative

[9] https://www.nifc.gov/fireInfo/fireInfo_documents/SuppCosts.pdf.

uncertainty associated with proactive versus reactive treatment alternatives. For example, in the context of changing wildfire regimes and climatic conditions, what are the uncertainties, trade-offs, and likely consequences to U.S., Canadian, and Mexican Indigenous and non-indigenous people, infrastructure, ecosystems, native species and habitats of

1) Restoring active fire regimes to dry, moist, and cold forest ecosystems,
2) Continued fire suppression in these same forest types,
3) Proposed proactive, reactive, and no-action management alternatives,
4) Post-fire forest regeneration under action and no-action alternatives, and
5) Post-fire harvest/non-harvest of the younger fire-killed trees to mimic reburns?

Reframing Management and Policy

As demonstrated in literature reviews, the disturbance ecology of an ecosystem may still be the most valuable lens through which climate-related events and outcomes may be understood (Long 2009, Newman 2019, Franklin and Johnson 2013). Over millennia, Indigenous fire use in many areas amplified fire frequency to reduce the likelihood of more extreme wildfires and their effects on culturally valued resources and conditions. This culturally modified disturbance regime increased the resilience and resistance of vegetation and landscape conditions to changing climatic conditions and associated disturbances.

More recently, ecological forestry principles recognize the value of management planning that incorporates the influence of natural disturbance processes on forest dynamics (Franklin et al. 2018). Additionally, as shown by Indigenous experience, natural lightning ignitions can be supplemented to achieve desired conditions. In uncertain times, management might better focus on the long-term persistence of that native biodiversity that evolved within the local, culturally enhanced, disturbance regime, and will likely go extinct with rapid or extreme changes to those regime properties (Newman 2019). Where possible, adapting local landscapes to conserve key aspects of culturally enhanced disturbance regimes could be vital to preserving functioning ecosystems and to the native biodiversity that requires non-extreme disturbance for its continued existence (Franklin and Johnson 2012, North et al. 2014), even where single-species conservation and broader ecosystem goals may appear to be in conflict at other scales (Henson et al. 2013).

Managers and scientists have repeatedly proposed management directions that incorporate knowledge of disturbance ecology and methods that adequately mimic and recover local disturbance regimes; however, socioeconomic challenges have impeded widespread implementation of these strategies (Long 2009). Effective direction would be proactive rather than reactive, recognizing that just as with human society, all desired ecological outcomes are not possible in the same place, at the same time.

Recommendations

In this light, what constitutes adaptation of wNA forests in these uncertain times? Is bias for action rather than inaction recommended?

Scientific knowledge is always growing and incomplete. However, a preponderance of evidence suggests that proactive management can prepare many landscapes for future wildfires and the maintenance work they can provide. This would also reduce emphasis on high-maintenance solutions and the overarching and increasingly burdensome role of wildfire suppression and its expenditures.

Emphasize whole landscape and multi-scale adaptation.—Stand management as applied to western U.S. and Canadian public lands typically emphasizes forested areas where there are commercial opportunities for mechanical treatments in specific stands of trees. This focus misses many locations where proactive treatments may be most useful to adapting an entire landscape. Conducting whole landscape evaluations of forest conditions, fire regime departures, and expected future climate and weather conditions can powerfully aid in defining those places that would most benefit from adaptation treatments (North et al. 2009, Hessburg et al. 2013, 2016, Meyer et al. 2021).

Ongoing collaborative partnerships also recommend that emphasis on timber volume production from public lands has a negative influence on partner support for projects and trust maintenance (Hessburg et al. 2020), and it tends to force the hand of managers to rank commercial treatments over others that may be more truly adaptive. Alternatively, management and planning that emphasizes area restored and adapted could build trust and a broader base of support, while still providing timber volume to sustain rural mills and economies (Rummer et al. 2005). Increasingly, collaborative restoration partnerships with Indigenous communities, having tribes as part of the leadership and management, can increase opportunities for reinstating tribal stewardship practices, with tribes, local communities, and the broader society as beneficiaries of active management that achieves shared values (Lake et al. 2018, Long and Lake 2018).

Large and old trees.—Most research reveals that broadly conserving large and old fire-resistant trees and replacing those that were removed or killed by harvest, drought, insects, pathogens, and wildfires provides a strong backbone of resilient structure and habitat to seasonally dry pine and mixed-conifer ecosystems (Spies et al. 2018, 2019).

Clumped and gapped trees.—As is appropriate to local seasonally dry forest types, restoring tree clump and gap patterns, and increasing area of these conditions will provide a solid patch-to-local landscape bet-hedging strategy in a warming future with increasing burned area (Larson and Churchill 2012, Churchill et al. 2013, LeFevre et al. 2020).

Successionally heterogeneous forests.—Successional heterogeneity, whether fine-, meso-, or coarse-grained, was an historical consequence of patterns of environmental productivity and fire–climate interactions with vegetation. It reinforced a continual shifting of diverse but similar patterns of heterogeneity at each scale of observation. As is appropriate to forest types and physiographic domains, restoring and maintaining forms of this heterogeneity will encourage a wider variety of wildfire and habitat outcomes, and reduce the need for aggressive fire suppression in many areas (Perry et al. 2011). This can be accomplished by adapting current spatial patterns of seral stages to more frequent burning and reburning (Stephens et al. 2020). Indigenous and Western knowledge can jointly aid in determining how best to adapt current and projected future conditions.

Using the topo-edaphic template.—Throughout wNA, topography, geomorphology, lithologies, and soils provide the template for spatially varying forest cover types, structural conditions, and their variations (Taylor and Skinner 2003, Hessburg et al. 2015). Fire exclusion and other influences have weakened connections to this template. Realigning spatial patterns of nonforest and forest successional pattern conditions (e.g., open vs. closed canopy, fire-tolerant vs. intolerant species) to this template will aid in adapting landscapes to changing climatic and wildfire regimes. For example, restoring non-forest and low-density forest cover patches to south-facing slopes and ridgetops and higher-density forest cover patches on north aspects and valley bottoms are examples of strengthening connections to the topographic template (Hessburg et al. 2015, 2019). These underlying conditions continually co-create the environments for disturbance and revegetation as the climate changes (Taylor and Skinner 2003, Hessburg et al. 2015, 2016).

Forests and their nonforests.—Meadows, shrublands, savannahs, and preforest conditions result from natural succession, disturbance dynamics, and reburning (Prichard et al. 2017). Restoring more characteristic nonforest-forest patterns in and among all forest types at fine, meso, and broad scales could significantly realign primary ecosystem processes, carbon storage, and hydrologic regimes with the warming climate (Shakesby and Doerr 2006). As recent history has shown, many pre-fire-suppression era nonforest areas throughout wNA became forested absent active fire regimes during the mild mid-20th-century climatic period (Hessburg et al. 2019).

Research Needs

The reviews of Hagmann et al. (2021) and Prichard et al. (2021) show deepening understanding of the fire and landscape ecology of wNA forests; however, substantive knowledge gaps remain. Here, we discuss the following research needs that emerged from this review:.

1) *Fortifying future vegetation and wildfire projections with insights from landscape ecology research* Most continental to regional projections of climate influence on biotic conditions and physical processes use a range of intuitive climatic drivers to explain responses to warming (Rosenzweig et al. 2008, Parks et al. 2015, Abatzoglou and Williams 2016). Outcomes are presented as ostensibly unaffected by bottom-up or meso-scale spatial variation in biotic, environmental, disturbance history, or topoedaphic conditions inherent to the system(s) of interest. From the standpoint of landscape ecology theory and practice, this approach misses key cross-scale interactions between the climate system and highly varying biophysical settings, which are known to modify climate system inputs and alter spatial and temporal patterns of realized conditions (Wu and Loucks 1995). Hurteau et al. (2019) for example, showed that future projections of burned area under climate change, which accounted for interactions among prior fires on surface and canopy fuel availability, reduced area burned by 14.3% in the Sierra Nevada compared to projections where only climate drivers were considered. Hybrid research and modeling are needed among climate change scientists and landscape ecologists to improve projections of vegetation and burned area changes, and species ranges.
2) *Multi-proxy evidence is more informative than single proxy* Observing and integrating knowledge of the multi-level dimensions of forest landscapes and their wildfire regimes provides deeper insight into how patterns influence processes, and it improves change detection (Hagmann et al. 2021). Some regions are already represented by multi-level studies, but in some cases, they could be better integrated. Multi-scale and multi-proxy historical reconstructions are still needed for other regions of wNA to better understand variations in forest–nonforest relations and successional heterogeneity that are better aligned with changing climatic and wildfire regimes. With these insights, managers and policy-makers will be better able to understand how warming and drying may affect adaptation strategies.
3) *More wildfire–forest dynamics carbon research is needed* Recent studies show that strategies for adapting current forests to wildfires and climate change may result in more terrestrial carbon storage than business-as-usual scenarios. The reason is that large fuel buildup under fire exclusion renders forest carbon stores vulnerable to large, high-severity fire

events (Liang et al. 2018, Hurteau et al. 2019). This nascent inquiry area deserves significant investment and increased scope to determine suitable landscape management strategies, and where they may best apply.

4) *Disturbance–fish-and-wildlife-habitat connections* As Newman (2019) suggests, native species and their habitats are tied to disturbance regimes of local and regional landscapes. Our current knowledge of these species-disturbance regime linkages is weak in many areas and could be much better understood. Even where 25 yr of research is available for one of the most intensively studied bird species, that knowledge is not preventing population declines for Northern Spotted Owls (Spies et al. 2019, Stephens et al. 2019). As a result, Henson et al. (2018) advocated a coarse-filter approach that incorporates disturbance ecology in management for spotted owl habitat. Understanding and managing spatial domains to restore these more functional disturbance regimes is an intuitive coarse-filter conservation strategy for terrestrial and aquatic species.

5) *A role for decision support tools* Predicting future vegetation, climate, and wildfire conditions, and trade-offs among various habitats and resources across a set of potential management scenarios is intellectually and computationally challenging. Considering the large number of data layers, the one-to-many and many-to-one interactions among conditions represented by these layers, and variation in these relationships by scenario thwarts careful evaluation by even the best planning intellects. Decision support tools are designed for this complex and integrated planning environment and are useful for evaluating trade-offs among changing conditions, outcomes, and management scenarios (Kangas and Kangas 2005, Reynolds et al. 2014). Using such tools, managers and scientists can observe trade-offs and related positive and negative cascades associated with varied management scenarios and discover their primary drivers.

6) *Innovation and investment in multi-party monitoring and adaptive management* Adaptive forest management has been recommended by scientists and managers for decades (Lee 1999), however, it has functioned more as an abstraction than an applied reality (Bormann et al. 2007). While adaptive management provided the core of Indigenous landscape management methods (Anderson 2013), there are several key reasons for delayed application in contemporary management. Adaptive management depends on watchful learning; what we today call ecosystem monitoring, which can be time consuming and expensive, and results often come after lengthy delays. Sufficient monitoring is rarely budgeted for, and consequently, an adaptive process is inhibited. Without agreement on the monitoring questions and goals of management, disputes remain unresolved. Innovation and investment are needed in this area to develop better methods of multi-party goal setting, and efficient and inexpensive means of monitoring; for example, multi-scale photography or remote sensing in addition to intensive plot and survey application. Another monitoring approach proposed by Tribes is to use cultural keystone species as indicators of ecosystem integrity and function (Garibaldi and Turner 2004). Results from monitoring a representative subset of forest conditions and projects could be extrapolated to similar conditions. This would enable more rapid learning and implementation, which are core concerns. Effective learning of this sort will become more essential as expanding human populations search for better ways to live sustainably on increasingly dynamic wNA landscapes.

Conclusions

Here, we have described how policy and management choices of the last two centuries yielded forest conditions throughout much of wNA that are vulnerable to the effects of rapid climatic warming, including increasing fire and drought severity. We summarized core messages of Hagmann et al. (2021) and Prichard et al. (2021), detailing widespread changes in forested landscapes and wildfire regimes since the influx of European colonists, and addressing popular questions about the capacity of management practices to reverse or mitigate the worst effects of these changes. We address concerns about the influence of agenda-driven science and reiterate that the *precautionary principle* can become the *paralyzing principle* given uncertainty about the risk of losses associated with action and no-action alternatives. We discussed the near impossibility of demonstrating lack of harm for most any action, including inaction, especially in a rapidly changing environment.

We provided recommendations for reframing forest and fire management and their related policy underpinnings, emphasizing (1) whole landscape and multi-scale adaptation; (2) protection of large and old fire- and drought-tolerant trees and old forests; (3) restoration of clumped and gapped tree patterns at fine and meso spatial scales; (4) creation of successionally heterogeneous forests; (5) use of topography to realign current conditions to the biophysical template; and (6) restoration of nonforest conditions. Climate change will create more nonforest and more young open canopy forest conditions (Parks et al. 2016, Hessburg et al. 2019, Coop et al. 2020); the opportunity for management is to place those conditions and patch sizes in locations that provide the greatest social and ecological benefits while conserving and recruiting old trees and old forest where possible.

Some today call for cultivating pyrodiversity to advance biodiversity (Parr and Andersen 2006, Taylor et al. 2012, Bowman et al. 2016). However, not all heterogeneity is equally well adapted to the topography, soils, and varied environmental settings and fire

regimes of wNA landscapes, and thus may endanger native biodiversity. Climate and wildfire adaptation requires structural and compositional patterns and pattern variations that are in synch with biophysical settings, reinforce the desired fire regimes, and reduce undesirable impacts of climatic warming to socioecological communities.

We close our review with a short list of research needs. Key among them is the need to better understand the disturbance regimes that native plants and animals evolved with and through which persistence occurred even as we act proactively to restore pattern–process interactions and adapt these landscapes to warming climate. Most legal battles concerning forest management today are about native biodiversity, old tree or old forest conservation, conservation of threatened and endangered species, and impacts of timber harvesting. Yet, native species and their habitats are tied to disturbance regimes of local and regional landscapes and their pattern variations. Our current knowledge of these species-disturbance regime linkages is weak, yet these dynamics might become a focal means of biodiversity conservation (Henson et al. 2013).

Finally, we discussed how some of these climatic and fire regime effects were common to landscapes inhabited by the Indigenous people of wNA, and in closing, we return to those ideas. Because of significant vulnerabilities linked to native wildfire regimes, Indigenous people intentionally managed wildfire for millennia to provide a broad variety of life-supporting resources, food and medicine security, protect lifeways, sacred places, and deeply held traditions, and to increase personal safety. This intentional management was a transgenerational commitment; prior generations took responsibility for the quality and abundance of desired conditions they passed on to subsequent generations. Since the mid-1850s, the majority of EuroAmerican colonists and present-day citizens have neither practiced this intentional management nor passed on a transgenerational commitment. Yet, we are ever more dependent as a society on the ecosystem services that functional fire-adapted landscapes provide. Given the known risks of modern wildfires and climate change, embracing the role of fire and a return to intentional transgenerational management is of critical importance.

Acknowledgments

The authors acknowledge constructive reviews by Keith Reynolds, Jamie Rosen, Derek Churchill, Gavin Jones, and two anonymous reviewers. We are grateful to David Schimel for the invitation to publish this small collection of papers, and to Brion Salter for cartography. We appreciate funding by the U.S. Fish and Wildlife Service, The Wilderness Society; The Nature Conservancy, Oregon; Conservation Northwest; The Ecological Restoration Institute; Washington State Department of Natural Resources; USDA-FS, the Pacific Northwest and Pacific Southwest Research Stations, and the California Department of Forestry and Fire Protection in support of these reviews.

Disclaimer

The findings and conclusions in this article are those of the author(s) and do not necessarily represent the views of the U.S. Fish and Wildlife Service or the USDA Forest Service.

Literature Cited

Abatzoglou, J. T., and A. P. Williams. 2016. Impact of anthropogenic climate change on wildfire across western US forests. Proceedings of the National Academy of Sciences USA 113:11770–11775.

Ager, A. A., A. M. Barros, R. Houtman, R. Seli, and M. A. Day. 2020. Modelling the effect of accelerated forest management on long-term wildfire activity. Ecological Modelling 421:108962. https://doi.org/10.1016/j.ecolmodel.2020.108962

Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, and J. T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications 12:1418–1433.

Anderson, M. K. 2013. Tending the wild: Native American knowledge and the management of California's natural resources. University of California Press, Berkeley, California, USA.

Asner, G. P., P. G. Brodrick, C. B. Anderson, N. Vaughn, D. E. Knapp, and R. E. Martin. 2016. Progressive forest canopy water loss during the 2012–2015 California drought. Proceedings of the National Academy of Sciences USA 113:E249–E255. https://doi.org/10.1073/pnas.1523397113

Balch, J. K., B. A. Bradley, J. T. Abatzoglou, R. C. Nagy, E. J. Fusco, and A. L. Mahood. 2017. Human-started wildfires expand the fire niche across the United States. Proceedings of the National Academy of Sciences of the United States of America 114:2946–2951.

Barnett, K., S. A. Parks, C. Miller, and H. T. Naughton. 2016. Beyond fuel treatment effectiveness: characterizing interactions between fire and treatments in the US. Forests 7:237. https://doi.org/10.3390/f7100237

Bastin, J. F., Y. Finegold, C. Garcia, D. Mollicone, M. Rezende, D. Routh, C. M. Zohner, and T. W. Crowther. 2019. The global tree restoration potential. Science 365:76–79.

Belsky, A., and D. Blumenthal. 1997. Effects of livestock grazing on stand dynamics and soils in upland forests of the interior west. Conservation Biology 11:315–327.

Bisson, P. A., B. E. Rieman, C. Luce, P. F. Hessburg, D. C. Lee, J. L. Kershner, G. H. Reeves, and R. E. Gresswell. 2003. Fire and aquatic ecosystems of the western USA: current knowledge and key questions. Forest Ecology and Management 178:213–229.

Bormann, B. T., R. W. Haynes, and J. R. Martin. 2007. Adaptive management of forest ecosystems: did some rubber hit the road? BioScience 57:186–191.

Bowman, D. M., G. L. Perry, S. I. Higgins, C. N. Johnson, S. D. Fuhlendorf, and B. P. Murphy. 2016. Pyrodiversity is the coupling of biodiversity and fire regimes in food webs. Philosophical Transactions of the Royal Society B 371:20150169.

Boyd, R. 1999. Indians, fire and the land. Pages 54. Oregon State University Press, Corvallis, Oregon, USA.

Bryant, T., K. Waring, A. Sánchez Meador, and J. B. Bradford. 2019. A framework for quantifying resilience to forest disturbance. Frontiers in Forests and Global Change 2:56. https://doi.org/10.3389/ffgc.2019.00056

Calkin, D. E., K. M. Gebert, J. G. Jones, and R. P. Neilson. 2005. Forest service large fire area burned and suppression expenditure trends, 1970–2002. Journal of Forestry 103:179–183.

Carey, A. B. 2006. Active and passive forest management for multiple values. Northwestern Naturalist 87:18–30. https://doi.org/10.1898/1051-1733(2006)87[18:AAPFMF]2.0.CO;2

Churchill, D. J., A. J. Larson, M. C. Dahlgreen, J. F. Franklin, P. F. Hessburg, and J. A. Lutz. 2013. Restoring forest resilience: from reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management 291:442–457.

Cohen, J. D. 2000. Preventing disaster: home ignitability in the wildland-urban interface. Journal of Forestry 98:15–21. https://doi.org/10.1093/jof/98.3.15

Coop, J. D., et al. 2020. Wildfire-driven forest conversion in Western North American landscapes. BioScience 70:659–673.

Coppoletta, M., K. E. Merriam, and B. M. Collins. 2016. Post-fire vegetation and fuel development influences fire severity patterns in reburns. Ecological Applications 26:686–699.

Dale, V. H., et al. 2001. Climate change and forest disturbances: climate change can affect forests by altering the frequency, intensity, duration, and timing of fire, drought, introduced species, insect and pathogen outbreaks, hurricanes, windstorms, ice storms, or landslides. BioScience 51:723–734. https://doi.org/10.1641/0006-3568(2001)051[0723:CCAFD]2.0.CO;2

Fettig, C. J., L. A. Mortenson, B. M. Bulaon, and P. B. Foulk. 2019. Tree mortality following drought in the central and southern Sierra Nevada, California, US. Forest Ecology and Management 432:164–178. https://doi.org/10.1016/j.foreco.2018.09.006

Franklin, J. F., and K. N. Johnson. 2012. A restoration framework for federal forests in the Pacific Northwest. Journal of Forestry 110:429–439.

Franklin, J. F., and K. N. Johnson. 2013. Ecologically based management: A future for federal forestry in the Pacific Northwest. Journal of Forestry 111:429–432.

Franklin, J. F., K. N. Johnson, and D. L. Johnson. 2018. Ecological forest management. Pages 646. Waveland Press, Long Grove, Illinois, USA.

Fulé, P. Z., et al. 2014. Unsupported inferences of high-severity fire in historical dry forests of the western United States: response to Williams and Baker. Global Ecology and Biogeography 23:825–830.

Garibaldi, A., and N. Turner. 2004. Cultural keystone species: implications for ecological conservation and restoration. Ecology and Society 9:1.

Garrard, G. E., F. Fidler, B. C. Wintle, Y. E. Chee, and S. A. Bekessy. 2016. Beyond advocacy: making space for conservation scientists in public debate. Conservation Letters 9:208–212.

Grainger, A., L. R. Iverson, G. H. Marland, and A. Prasad. 2019. Comment on "The global tree restoration potential". Science 366:eaay8334.

Hagmann, R. K., et al. 2021. Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests. Ecological Applications e02431. https://doi.org/10.1002/eap.2431

Hagmann, R. K., J. F. Franklin, and K. N. Johnson. 2013. Historical structure and composition of ponderosa pine and mixed-conifer forests in south-central Oregon. Forest Ecology and Management 304:492–504.

Hagmann, R. K., J. F. Franklin, and K. N. Johnson. 2014. Historical conditions in mixed-conifer forests on the eastern slopes of the northern Oregon Cascade Range, USA. Forest Ecology and Management 330:158–170.

Henson, P., J. Thrailkill, B. Glenn, B. Woodbridge, and B. White. 2013. Using ecological forestry to reconcile spotted owl conservation and forest management. Journal of Forestry 111:433–437.

Henson, P., R. White, and S. P. Thompson. 2018. Improving implementation of the Endangered Species Act: finding common ground through common sense. BioScience 68:861–872. https://doi.org/10.1093/biosci/biy093

Hessburg, P. F., et al. 2015. Restoring fire-prone Inland Pacific landscapes: seven core principles. Landscape Ecology 30:1805–1835.

Hessburg, P. F., et al. 2016. Tamm review: management of mixed-severity fire regime forests in Oregon, Washington, and Northern California. Forest Ecology and Management 366:221–250.

Hessburg, P. F., et al. 2019. Climate, environment, and disturbance history govern resilience of western North American forests. Frontiers in Ecology and Evolution 7:239. https://doi.org/10.3389/fevo.2019.00239

Hessburg, P. F., et al. 2020. The 1994 Eastside Screens large-tree harvest limit: review of science relevant to forest planning 25 years later. Pages 114. General Technical Report PNW-GTR-990. US Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Hessburg, P. F., and J. K. Agee. 2003. An environmental narrative of inland northwest United States forests, 1800–2000. Forest Ecology and Management 178:23–59.

Hessburg, P. F., J. K. Agee, and J. F. Franklin. 2005. Dry forests and wildland fires of the inland Northwest USA: Contrasting the landscape ecology of the pre-settlement and modern eras. Forest Ecology and Management 211:117–139.

Hessburg, P. F., K. M. Reynolds, R. B. Salter, J. D. Dickinson, W. L. Gaines, and R. J. Harrod. 2013. Landscape evaluation for restoration planning on the Okanogan-Wenatchee National Forest, USA. Sustainability 5:805–840. https://doi.org/10.3390/su5030805

Hessburg, P. F., B. G. Smith, and R. B. Salter. 1999. Detecting change in forest spatial patterns from reference conditions. Ecological Applications 9:1232–1252.

Hessburg, P. F., B. G. Smith, R. B. Salter, R. D. Ottmar, and E. Alvarado. 2000. Recent changes (1930s–1990s) in spatial patterns of interior northwest forests, USA. Forest Ecology and Management 136:53–83. https://doi.org/10.1016/S0378-1127(99)00263-7

Higuera, P. E., M. E. Peters, L. B. Brubaker, and D. G. Gavin. 2007. Understanding the origin and analysis of sediment-charcoal records with a simulation model. Quaternary Science Reviews 26:1790–1809. https://doi.org/10.1016/j.quascirev.2007.03.010

Hof, A. R., C. C. Dymond, and D. J. Mladenoff. 2017. Climate change mitigation through adaptation: the effectiveness of forest diversification by novel tree planting regimes. Ecosphere 8:e01981.

Huffman, M. R. 2013. The many elements of traditional fire knowledge: synthesis, classification, and aids to cross-cultural problem solving in fire-dependent systems around the world. Ecology and Society 18:3.

Hurteau, M. D., J. B. Bradford, P. Z. Fulé, A. H. Taylor, and K. L. Martin. 2014. Climate change, fire management, and ecological services in the southwestern US. Forest Ecology and Management 327:280–289.

Hurteau, M. D., S. Liang, A. L. Westerling, and C. Wiedinmyer. 2019. Vegetation-fire feedback reduces projected area burned under climate change. Scientific Reports 9:1–6.

Isaak, D. J., C. H. Luce, B. E. Rieman, D. E. Nagel, E. E. Peterson, D. L. Horan, S. Parkes, and G. L. Chandler. 2010. Effects of climate change and wildfire on stream temperatures

and salmonid thermal habitat in a mountain river network. Ecological Applications 20:1350–1371.

Kangas, J., and A. Kangas. 2005. Multiple criteria decision support in forest management—the approach, methods applied, and experiences gained. Forest Ecology and Management 207:133–143. https://doi.org/10.1016/j.foreco.2004.10.023

Kay, C. E. 2000. Native burning in western North America: implications for hardwood forest management. Pages 19–27 *in* A. D. Yaussy, comp. Proceedings: Workshop on Fire, People, and the Central Hardwoods Landscape. General Technical Report NE-274. US Department of Agriculture, Forest Service, Northeastern Research Station, Newtown Square, Pennsylvania, USA.

Keane, R. E., G. J. Cary, I. D. Davies, M. D. Flannigan, R. H. Gardner, S. Lavorel, J. M. Lenihan, C. Li, and T. S. Rupp. 2004. A classification of landscape fire succession models: spatial simulations of fire and vegetation dynamics. Ecological Modelling 179:3–27. https://doi.org/10.1016/j.ecolmodel.2004.03.015

Keane, R. E., R. A. Loehman, L. M. Holsinger, D. A. Falk, P. Higuera, S. M. Hood, and P. F. Hessburg. 2018. Use of landscape simulation modeling to quantify resilience for ecological applications. Ecosphere 9:e02414.

Keane, R. E., K. C. Ryan, T. T. Veblen, C. D. Allen, J. A. Logan, and B. Hawkes. 2002. The cascading effects of fire exclusion in Rocky Mountain ecosystems, Chapter 7. Pages 133–152 *in* J. Baron, editor. Rocky Mountain futures: An ecological perspective. Island Press, Washington, D.C., USA.

Kolden, C. A. 2019. We're not doing enough prescribed fire in the Western United States to mitigate wildfire risk. Fire 2:30. https://doi.org/10.3390/fire2020030

Kolden, C. A., and C. Henson. 2019. A socio-ecological approach to mitigating wildfire vulnerability in the wildland urban interface: a case study from the 2017 Thomas Fire. Fire 2:9.

Komatsu, H., and T. Kume. 2020. Modeling of evapotranspiration changes with forest management practices: A genealogical review. Journal of Hydrology 585:124835.

Lach, D., P. List, B. Steel, and B. Shindler. 2003. Advocacy and credibility of ecological scientists in resource decision-making: a regional study. BioScience 53:170–178. https://doi.org/10.1641/0006-3568(2003)053[0170:AACOES]2.0.CO;2

Lackey, R. T. 2007. Science, scientists, and policy advocacy. Conservation Biology 21:12–17.

Lake, F. K., and A. C. Christianson. 2019. Indigenous fire stewardship. Pages 1–9 *in* S. Mazello, editor. Encyclopedia of wildfire and wildland-urban interface (WUI) fires. Springer, Cham, Switzerland. https://doi.org/10.1007/978-3-319-51727-8_225-1

Lake, F. K., J. Parrotta, C. P. Giardina, I. Davidson-Hunt, and Y. Uprety. 2018. Integration of traditional and Western knowledge in forest landscape restoration. Pages 214–242 *in* S. Mansourian and J. Parrotta, editors. Forest landscape restoration: integrated approaches to support effective implementation. Routledge, Abingdon-on-Thames, UK.

Larson, A. J., and D. Churchill. 2012. Tree spatial patterns in fire-frequent forests of western North America, including mechanisms of pattern formation and implications for designing fuel reduction and restoration treatments. Forest Ecology and Management 267:74–92.

Lee, K. N. 1999. Appraising adaptive management. Conservation Ecology 3:1–21.

LeFevre, M. E., D. J. Churchill, A. J. Larson, S. M. Jeronimo, J. Bass, J. F. Franklin, and V. R. Kane. 2020. Evaluating restoration treatment effectiveness through a comparison of residual composition, structure, and spatial pattern with historical reference sites. Forest Science 66:578–588.

Levine, C. R., C. V. Cogbill, B. M. Collins, A. J. Larson, J. A. Lutz, M. P. North, C. M. Restaino, H. D. Safford, S. L. Stephens, and J. J. Battles. 2017. Evaluating a new method for reconstructing forest conditions from General Land Office survey records. Ecological Applications 27:1498–1513.

Lewis, S. L., E. T. Mitchard, C. Prentice, M. Maslin, and B. Poulter. 2019. Comment on "The global tree restoration potential". Science 366:eaaz0388.

Liang, S., M. D. Hurteau, and A. L. Westerling. 2018. Large-scale restoration increases carbon stability under projected climate and wildfire regimes. Frontiers in Ecology and the Environment 16:207–212.

Lightfoot, K. G., et al. 2013. Anthropogenic burning on the central California coast in Late Holocene and early historical times: findings, implications, and future directions. California Archaeology 5:371–390.

Long, J. N. 2009. Emulating natural disturbance regimes as a basis for forest management: a North American view. Forest Ecology and Management 257:1868–1873.

Long, J. W., and F. K. Lake. 2018. Escaping social-ecological traps through tribal stewardship on national forest lands in the Pacific Northwest, United States of America. Ecology and Society 23:10.

Long, J. W., F. K. Lake, R. W. Goode, and B. M. Burnette. 2020. How traditional tribal perspectives influence ecosystem restoration. Ecopsychology 12:71–82.

Lydersen, J. M., M. P. North, E. E. Knapp, and B. M. Collins. 2013. Quantifying spatial patterns of tree groups and gaps in mixed-conifer forests: Reference conditions and long-term changes following fire suppression and logging. Forest Ecology and Management 304:370–382.

Madeira, L., and T. Gartner. 2018. Forest resilience bond sparks innovative collaborations between water utilities and wide-ranging stakeholders. Journal American Water Works Association 110:42–49. http://doi: 10.1002/awwa.1097

McGarigal, K., and W. H. Romme. 2012. Modeling historical range of variation at a range of scales: an example application. Pages 128–145 *in* J. Wiens, C. Regan, G. Hayward, and H. Safford, editors. Historical environmental variation in conservation and natural resource management. Wiley, New York, New York, USA.

McKenzie, D., Z. E. Gedalof, D. L. Peterson, and P. Mote. 2004. Climatic change, wildfire, and conservation. Conservation Biology 18:890–902.

Meyer, M. D., J. W. Long, and H. D. Safford. 2021. Postfire restoration framework for national forests in California. Pages 204. General Technical Report PSW-GTR-270. US Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Mildrexler, D. J., L. T. Berner, B. E. Law, R. A. Birdsey, and W. R. Moomaw. 2020. Large trees dominate carbon storage in forests east of the cascade crest in the United States Pacific Northwest. Frontiers in Forests and Global Change 3. http://dx.doi.org/10.3389/ffgc.2020.594274

Millar, C. I., N. L. Stephenson, and S. L. Stephens. 2007. Climate change and forests of the future: managing in the face of uncertainty. Ecological Applications 17:2145–2151.

Moreira, F., et al. 2020. Wildfire management in Mediterranean-type regions: paradigm change needed. Environmental Research Letters 15:011001.

Morgan, P., E. K. Heyerdahl, and C. E. Gibson. 2008. Multi-season climate synchronized forest fires throughout the 20th century, northern Rockies, USA. Ecology 89:717–728.

Moritz, M. A., M. D. Hurteau, K. N. Suding, and C. M. D'Antonio. 2013. Bounded ranges of variation as a framework for future conservation and fire management. Annals of the New York Academy of Sciences 1286:92–107.

Newman, E. A. 2019. Disturbance ecology in the Anthropocene. Frontiers in Ecology and Evolution 7:147.

Norgaard, K. M. 2014. The politics of fire and the social impacts of fire exclusion on the Klamath. Humboldt Journal of Social Relations 36:77–101.

North, M., B. Collins, J. Keane, J. W. Long, C. Skinner, and B. Zielinski. 2014. Synopsis of emergent approaches. Pages 55–70 *in* J. W. Long, L. Quinn-Davidson, and C. N. Skinner, editors. Science synthesis to support socioecological resilience in the Sierra Nevada and southern Cascade Range. General Technical Report PSW-GTR-247. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California, USA.

North, M. P., S. L. Stephens, B. M. Collins, J. K. Agee, G. Aplet, J. F. Franklin, and P. Z. Fule. 2015. Reform forest fire management. Science 349:1280–1281.

North, M., P. Stine, K. O'Hara, W. Zielinski, and S. Stephens. 2009. An ecosystem management strategy for Sierran mixed-conifer forests. Pages 49. General Technical Report PSW-GTR-220 (Second printing, with addendum). US Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Noss, R. F., J. F. Franklin, W. L. Baker, T. Schoennagel, and P. B. Moyle. 2006. Managing fire-prone forests in the western United States. Frontiers in Ecology and the Environment 4:481–487.

Odion, D. C., et al. 2014. Examining historical and current mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. PLoS ONE 9:e87852.

Parks, S. A., and J. T. Abatzoglou. 2020. Warmer and drier fire seasons contribute to increases in area burned at high severity in western US forests from 1985 to 2017. Geophysical Research Letters 47:e2020GL089858. https://doi.org/10.3389/ffgc.2021.629198

Parks, S. A., C. Miller, J. T. Abatzoglou, L. M. Holsinger, M. A. Parisien, and S. Z. Dobrowski. 2016. How will climate change affect wildland fire severity in the western US? Environmental Research Letters 11:035002.

Parks, S. A., C. Miller, M. A. Parisien, L. M. Holsinger, S. Z. Dobrowski, and J. Abatzoglou. 2015. Wildland fire deficit and surplus in the western United States, 1984–2012. Ecosphere 6:1–13. https://doi.org/10.1890/ES15-00294.1

Parr, C. L., and A. N. Andersen. 2006. Patch mosaic burning for biodiversity conservation: a critique of the pyrodiversity paradigm. Conservation Biology 20:1610–1619.

Peery, M. Z., G. M. Jones, R. J. Gutiérrez, S. M. Redpath, A. B. Franklin, D. Simberloff, M. G. Turner, V. C. Radeloff, and G. C. White. 2019. The conundrum of agenda-driven science in conservation. Frontiers in Ecology and the Environment 17:80–82. http://doi:10.1002/fee.2006

Perry, D. A., P. F. Hessburg, C. N. Skinner, T. A. Spies, S. L. Stephens, A. H. Taylor, J. F. Franklin, B. McComb, and G. Riegel. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. Forest Ecology and Management 262:703–717.

Pielke, R. 2007. The honest broker: making sense of science in policy and politics. Cambridge University Press, London, UK.

Povak, N. A., P. F. Hessburg, and R. B. Salter. 2018. Evidence for scale-dependent topographic controls on wildfire spread. Ecosphere 9:e02443.

Power, M. J., B. F. Codding, A. H. Taylor, T. W. Swetnam, K. E. Magargal, D. W. Bird, and J. F. O'Connell. 2018. Human fire legacies on ecological landscapes. Frontiers in Earth Science 6:151.

Prichard, S. J., et al. 2021. Adapting western US forests to wildfires and climate change: ten common questions. Ecological Applications. http://dx.doi.org/10.1002/eap.2433

Prichard, S. J., C. S. Stevens-Rumann, and P. F. Hessburg. 2017. Tamm review: shifting global fire regimes: lessons from reburns and research needs. Forest Ecology and Management 396:217–233.

Radeloff, V. C., et al. 2018. Rapid growth of the US wildland-urban interface raises wildfire risk. Proceedings of the National Academy of Sciences USA 115:3314–3319.

Raphael, M. G., M. J. Wisdom, M. M. Rowland, R. S. Holthausen, B. C. Wales, B. G. Marcot, and T. D. Rich. 2001. Status and trends of habitats of terrestrial vertebrates in relation to land management in the interior Columbia River Basin. Forest Ecology and Management 153:63–87.

Reynolds, K. M., P. F. Hessburg, and P. S. Bourgeron. 2014. Making transparent environmental management decisions. Springer, Berlin and Heidelberg, Germany.

Rieman, B. E., P. F. Hessburg, C. Luce, and M. R. Dare. 2010. Wildfire and management of forests and native fishes: conflict or opportunity for convergent solutions? BioScience 60:460–468.

Rosenzweig, C., et al. 2008. Attributing physical and biological impacts to anthropogenic climate change. Nature 453:353–357.

Rummer, B., et al. 2005. A strategic assessment of forest biomass and fuel reduction treatments in Western States. Pages 17. General Technical Report RMRS-GTR-149. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Schoennagel, T., et al. 2017. Adapt to more wildfire in western North American forests as climate changes. Proceedings of the National Academy of Sciences USA 114:4582–4590.

Schoennagel, T., R. L. Sherriff, and T. T. Veblen. 2011. Fire history and tree recruitment in the Colorado Front Range upper montane zone: implications for forest restoration. Ecological Applications 21:2210–2222.

Scott, J. M., J. L. Rachlow, R. T. Lackey, A. B. Pidgorna, J. L. Aycrigg, G. R. Feldman, L. K. Svancara, D. A. Rupp, D. I. Stanish, and R. K. Steinhorst. 2007. Policy advocacy in science: prevalence, perspectives, and implications for conservation biologists. Conservation Biology 21:29–35.

Shakesby, R. A., and S. H. Doerr. 2006. Wildfire as a hydrological and geomorphological agent. Earth-Science Reviews 74:269–307.

Soulé, M. E. 1985. What is conservation biology? BioScience 35:727–734. https://doi.org/10.2307/1310054

Sowerwine, J., D. Sarna-Wojcicki, M. Mucioki, L. Hillman, F. K. Lake, and E. Friedman. 2019*a*. Enhancing indigenous food sovereignty: a five-year collaborative tribal-university research and extension project in California and Oregon. Journal of Agriculture, Food Systems, and Community Development 9:167–190. https://doi.org/10.5304/jafscd.2019.09B.013

Sowerwine, J., M. Mucioki, D. Sarna-Wojcicki, and L. Hillman. 2019*b*. Reframing food security by and for Native American communities: a case study among tribes in the Klamath River basin of Oregon and California. Food Security 11:579–607. https://doi.org/10.1007/s12571-019-00925-y

Spies, T. A., et al. 2014. Examining fire-prone forest landscapes as coupled human and natural systems. Ecology and Society 19:9. http://dx.doi.org/10.5751/ES-06584-190309

Spies, T. A., et al. 2019. Twenty-five years of the Northwest Forest Plan: what have we learned? Frontiers in Ecology and the Environment 17:511–520.

Spies, T. A., P. F. Hessburg, C. N. Skinner, K. J. Puettmann, M. J. Reilly, R. J. Davis, J. A. Kertis, J. W. Long, and D. C. Shaw. 2018. Old growth, disturbance, forest succession, and management in the area of the Northwest Forest Plan. Pages 95–243, 966 *in* T. A. Spies, P. A. Stine, R. Gravenmier, J. W. Long, and M. J. Reilly, tech. coords. Synthesis of science to inform

land management within the Northwest Forest Plan area. General Technical Report PNW-GTR-966. US Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Stephens, S. L., et al. 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western US forests. Ecological Applications 19:305–320. https://doi.org/10.1890/07-1755.1

Stephens, S. L., et al. 2019. Is fire “for the birds”? How two rare species influence fire management across the US. Frontiers in Ecology and the Environment 17:391–399. https://doi.org/10.1002/fee.2076

Stephens, S. L., C. I. Millar, and B. M. Collins. 2010. Operational approaches to managing forests of the future in Mediterranean regions within a context of changing climates. Environmental Research Letters 5:024003.

Stephens, S. L., B. M. Collins, C. J. Fettig, M. A. Finney, C. M. Hoffman, E. E. Knapp, M. P. North, H. Safford, and R. B. Wayman. 2018. Drought, tree mortality, and wildfire in forests adapted to frequent fire. BioScience 68:77–88.

Stephens, S. L., A. L. Westerling, M. D. Hurteau, M. Z. Peery, C. A. Schultz, and S. Thompson. 2020. Fire and climate change: conserving seasonally dry forests is still possible. Frontiers in Ecology and the Environment 18:354–360.

Stevens, J. T., H. D. Safford, and A. M. Latimer. 2014. Wildfire-contingent effects of fuel treatments can promote ecological resilience in seasonally dry conifer forests. Canadian Journal of Forest Research 44:843–854.

Stevens-Rumann, C., and P. Morgan. 2016. Repeated wildfires alter forest recovery of mixed-conifer ecosystems. Ecological Applications 26:1842–1853.

Stevens, J. T., et al. 2016. Average stand age from forest inventory plots does not describe historical fire regimes in ponderosa pine and mixed-conifer forests of western North America. PLoS One 11:e0147688.

Stewart, O. C. 2002. Forgotten fires: native Americans and the transient wilderness. University of Oklahoma Press, Norman, Oklahoma, USA.

Stockdale, C. A., C. Bozzini, S. E. MacDonald, and E. Higgs. 2015. Extracting ecological information from oblique angle terrestrial landscape photographs: performance evaluation of the WSL Monoplotting Tool. Applied Geography 63:315–325. https://doi.org/10.1016/j.apgeog.2015.07.012

Stockdale, C. A., S. E. MacDonald, and E. Higgs. 2019. Forest closure and encroachment at the grassland interface: a century-scale analysis using oblique repeat photography. Ecosphere 10:e02774. https://doi.org/10.1002/ecs2.2774

Sunstein, C. 2003. The paralyzing principle: Does the precautionary principle point us in any helpful direction?. Regulation, Winter, 2002–2003.

Syphard, A. D., J. E. Keeley, A. B. Massada, T. J. Brennan, and V. C. Radeloff. 2012. Housing arrangement and location determine the likelihood of housing loss due to wildfire. PLoS ONE 7:e33954.

Taylor, A. H., and C. N. Skinner. 2003. Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecological Applications 13:704–719.

Taylor, A. H., V. Trouet, C. N. Skinner, and S. L. Stephens. 2016. Socioecological transitions trigger fire regime shifts and modulate fire–climate interactions in the Sierra Nevada, USA, 1600–2015 CE. Proceedings of the National Academy of Sciences USA 113:13684–13689.

Taylor, R. S., S. J. Watson, D. G. Nimmo, L. T. Kelly, A. F. Bennett, and M. F. Clarke. 2012. Landscape-scale effects of fire on bird assemblages: does pyrodiversity beget biodiversity? Diversity and Distributions 18:519–529.

Thomas, D., D. Butry, S. Gilbert, D. Webb, and J. Fung. 2017. The costs and losses of wildfires. NIST Special Publication 1215. http://dx.doi.org/10.6028/NIST.SP.1215

Vaillant, N. M., and E. D. Reinhardt. 2017. An evaluation of the Forest Service Hazardous Fuels Treatment Program—Are we treating enough to promote resiliency or reduce hazard? Journal of Forestry 115:300–308.

Varghese, J., and S. S. Crawford. 2021. A cultural framework for Indigenous, Local, and Science knowledge systems in ecology and natural resource management. Ecological Monographs 91:e01431.

Veldman, J. W., et al. 2019. Comment on “The global tree restoration potential”. Science 366:eaay7976.

Wei, Y., M. P. Thompson, J. H. Scott, C. D. O’Connor, and C. J. Dunn. 2019. Designing operationally relevant daily large fire containment strategies using risk assessment results. Forests 10:311. https://doi.org/10.3390/f10040311

Wellerstein, A. 2018. The myth of apolitical science. Science 362:1006. http://doi.org/ 10.1126/science.aav4900

Westerling, A. L. 2016. Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. Philosophical Transactions of the Royal Society B 371:20150178.

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increase western US forest wildfire activity. Science 313:940–943. http://doi.org/10.1126/science.1128834

Westerling, A. L., M. G. Turner, E. A. H. Smithwick, W. H. Romme, and M. G. Ryan. 2011. Continued warming could transform Greater Yellowstone fire regimes by mid-21st century. Proceedings of the National Academy of Sciences USA 108:13165–13170.

Williams, M. A., and W. L. Baker. 2012. Spatially extensive reconstructions show variable-severity fire and heterogeneous structure in historical western United States dry forests. Global Ecology and Biogeography 21:1042–1052.

Wood, C. M., and G. M. Jones. 2019. Framing management of social-ecological systems in terms of the cost of failure: the Sierra Nevada, USA as a case study. Environmental Research Letters 14:105004.

Wu, J., and O. L. Loucks. 1995. From balance of nature to hierarchical patch dynamics: a paradigm shift in ecology. Quarterly Review of Biology 70:439–466. https://doi.org/10.1086/419172

Ecological Applications, 31(8), 2021, e02433

Adapting western North American forests to climate change and wildfires: 10 common questions

Susan J. Prichard,[1,19] Paul F. Hessburg,[1,2] R. Keala Hagmann,[1,3] Nicholas A. Povak,[4] Solomon Z. Dobrowski,[5] Matthew D. Hurteau,[6] Van R. Kane,[1] Robert E. Keane,[7] Leda N. Kobziar,[8] Crystal A. Kolden,[9] Malcolm North,[10] Sean A. Parks,[11] Hugh D. Safford,[12] Jens T. Stevens,[13] Larissa L. Yocom,[14] Derek J. Churchill,[15] Robert W. Gray,[16] David W. Huffman,[17] Frank K. Lake,[18] and Pratima Khatri-Chhetri[1]

[1]*University of Washington School of Environmental and Forest Sciences, Seattle, Washington 98195-2100 USA*
[2]*U.S. Forest Service PNW Research Station, Wenatchee, Washington 98801 USA*
[3]*Applegate Forestry LLC, Corvallis, Oregon 97330 USA*
[4]*U.S. Forest Service, Pacific Southwest Research Station, Institute of Forest Genetics, 2480 Carson Road, Placerville, California 95667 USA*
[5]*University of Montana College of Forestry and Conservation, Missoula, Montana 59812 USA*
[6]*University of New Mexico Biology Department, Albuquerque, New Mexico 87131-0001 USA*
[7]*U.S. Forest Service Rocky Mountain Research Station, Missoula Fire Sciences Laboratory, Missoula, Montana 59808 USA*
[8]*Department of Natural Resources and Society, University of Idaho, Moscow, Idaho 83844 USA*
[9]*School of Engineering, University of California Merced, Merced, California 95343 USA*
[10]*U.S. Forest Service Pacific Southwest Research Station, 1731 Research Park, Davis, California 95618 USA*
[11]*U.S. Forest Service Aldo Leopold Wilderness Research Institute, Missoula, Montana 59801 USA*
[12]*U.S. Forest Service Pacific Southwest Research Station, Albany, California 94710 USA*
[13]*U.S. Geological Survey Fort Collins Science Center, New Mexico Landscapes Field Station, Santa Fe, New Mexico 87544 USA*
[14]*Department of Wildland Resources and Ecology Center, Utah State University College of Agriculture and Applied Sciences, Logan, Utah 84322 USA*
[15]*Washington State Department of Natural Resources Forest Health Program, Olympia, Washington 98504 USA*
[16]*R.W. Gray Consulting, Chilliwack, British Columbia V2R2N2 Canada*
[17]*Northern Arizona University Ecological Restoration Institute, Flagstaff, Arizona 86011 USA*
[18]*U.S. Forest Service Pacific Southwest Research Station, Arcata, California 95521 USA*

Citation: Prichard, S. J., et al. 2021. Adapting western North American forests to climate change and wildfires: 10 common questions. Ecological Applications 31(8):e02433. 10.1002/eap.2433

Abstract. We review science-based adaptation strategies for western North American (wNA) forests that include restoring active fire regimes and fostering resilient structure and composition of forested landscapes. As part of the review, we address common questions associated with climate adaptation and realignment treatments that run counter to a broad consensus in the literature. These include the following: (1) Are the effects of fire exclusion overstated? If so, are treatments unwarranted and even counterproductive? (2) Is forest thinning alone sufficient to mitigate wildfire hazard? (3) Can forest thinning and prescribed burning solve the problem? (4) Should active forest management, including forest thinning, be concentrated in the wildland urban interface (WUI)? (5) Can wildfires on their own do the work of fuel treatments? (6) Is the primary objective of fuel reduction treatments to assist in future firefighting response and containment? (7) Do fuel treatments work under extreme fire weather? (8) Is the scale of the problem too great? Can we ever catch up? (9) Will planting more trees mitigate climate change in wNA forests? And (10) is post-fire management needed or even ecologically justified? Based on our review of the scientific evidence, a range of proactive management actions are justified and necessary to keep pace with changing climatic and wildfire regimes and declining forest heterogeneity after severe wildfires. Science-based adaptation options include the use of managed wildfire, prescribed burning, and coupled mechanical thinning and prescribed burning as is consistent with land management allocations and forest conditions. Although some current models of fire management in wNA are averse to short-term risks and uncertainties, the long-term environmental,

Manuscript received 18 November 2020; revised 9 March 2021; accepted 22 March 2021; final version received 4 July 2021. Corresponding Editor: David S. Schimel.
[19] E-mail: sprich@uw.eu

social, and cultural consequences of wildfire management primarily grounded in fire suppression are well documented, highlighting an urgency to invest in intentional forest management and restoration of active fire regimes.

Key words: adaptive management; carbon; climate change; Climate Change and Western Wildfires; cultural burning; ecological resilience; forest management; fuel treatments; managed wildfire; mechanical thinning; prescribed fire; restoration; wildland fire.

Introduction

Forested landscapes across much of western North America (wNA) are significantly departed from historical structure, species composition, and wildland fire regime characteristics (Hagmann et al. 2021), and as such, their resilience and resistance to rapidly changing wildfire and climatic regimes are compromised (Stephens et al. 2020, Hessburg et al. 2021). Through a variety of causes, including curtailment of Indigenous burning practices, livestock grazing, and modern fire suppression, fire frequency in the 20th century decreased in many wNA forests (Marlon et al. 2012, Hessburg et al. 2019). The absence of fire and past forest management have led to profound changes in ecosystem structure, composition, and processes over the last two centuries (Hessburg et al. 2005, Parks et al. 2015*b*, Haugo et al. 2019). As the climate warms, forested landscapes face increasing vulnerability to rapid and extensive ecosystem changes from severe, large-scale disturbances such as persistent droughts, insect outbreaks, disease epidemics, and high-severity fires (Allen et al. 2010, Bentz et al. 2010, Crockett and Westerling 2017).

Historically, wildland fires, including human and lightning ignitions, varied in size, intensity, duration, and seasonality (Perry et al. 2011, Hessburg et al. 2016). Patterns of burning and re-burning created mosaics of severity, species distributions, and resource conditions within shifting patchworks of forest and nonforest vegetation and fuels, thereby limiting the extent of stand-replacing fire events (Prichard et al. 2017, Nigro and Molinari 2019, Hagmann et al. 2021). In the context of fire exclusion and climate change, many fire-prone forests now exhibit high surface, ladder, and canopy fuel contagion with lasting implications for ecosystem changes, carbon storage, hydrologic regimes, native biodiversity, and terrestrial and aquatic habitats (Ager et al. 2007, Coop et al. 2020).

In recent decades, increased area burned by western wildfires has been associated with uncharacteristically large patches of high-severity, stand-replacing fire (Parks and Abatzoglou 2020, Hagmann et al. 2021). In some regions, such as the Sierra Nevada Range in California and eastern Cascades of Washington state, area burned by high-severity fire is 4–10 times that of historical fire regimes (Mallek et al. 2013, Reilly et al. 2017). Because high-severity fire events can be catalysts for vegetation change, particularly when coupled with warmer and drier climatic conditions, trends in large wildfires and burn severity have implications for rapid ecosystem shifts and declines in valued resources (Kemp et al. 2019, Stevens-Rumann and Morgan 2019, Coop et al. 2020).

There is growing awareness of the vulnerability of many wNA forests and human communities to changing wildfire and climatic regimes (North et al. 2015*b*, Hessburg et al. 2016). Under the United States National Cohesive Wildland Fire Management Strategy (United States Department of Agriculture and United States Department of Interior 2021), multi-entity, cross-jurisdictional partnerships have formed to increase the pace and scale of forest adaptation and restorative treatments to promote broad-based landscape resilience to fire, fire-adapted communities, and safe and effective wildfire responses. Similarly, recent large wildfires (>1.2 million ha in both 2017 and 2018) in western Canada are prompting re-examination of forest fire management practices and the need to restore more fire-resilient landscapes (Parisien et al. 2020, Tymstra et al. 2020). Northern Mexico and Baja peninsula forests have experienced a much shorter period of fire exclusion, but a growing fire deficit mirrors trends in the United States and Canada (Rivera-Huerta et al. 2016, Yocom Kent et al. 2017).

Over the past two decades, there has been confusion in some of the scientific literature and popular media surrounding changes in the nature and extent of forest and fire regime changes (Hagmann et al. 2021), and the need for and efficacy of adaptation or restorative treatments. Since some treatments can involve the commercial sale of timber, they can be viewed through the lens of conflict over the role of timber production on federal, tribal and private forestlands. The legacy of mistrust from these conflicts affects how different groups perceive the science and its application in support of proactive efforts to increase the resilience of forested landscapes (Schultz and Jedd 2012, Dubay et al. 2013). Perceived uncertainty in the science of fuel treatments and adaptive forest management has the potential to hinder collaborative decision-making, weaken public support for adaptive forest management, and slow implementation of needed forest management, particularly where courts rule that the science is yet unsettled. For example, in a recent opinion on a proposed forest restoration project, US State Court of Appeals for the Ninth Circuit Judge Graber wrote, “The project’s proposed methodology of variable density thinning is both highly controversial and highly uncertain.” (*BARK* et al. *v. U.S. Forest Service.* No. 3:18-cv-01645-MO). Given current warming trends, changing wildfire regimes, and climate projections for the balance of this century, the current slow pace and small scale of adaptive management

portend that many forest landscapes will experience uncharacteristic, high-severity wildfires and/or insect outbreaks before treatments can occur (North et al. 2015*b*, McWethy et al. 2019). High-severity disturbance events often have long-lasting impacts, including losses to ecosystem services and valued resources, shifts to new ecosystem types, and reduced options for future adaptation (Stevens-Rumann and Morgan 2019).

Under climate change, land development, and the spread of invasive species, adaptive forest management is not intended to return systems to historical reference conditions (Allen et al. 2011, Falk et al. 2019). Nonetheless, adaptive strategies prompt managers to define a set of historical and future reference conditions that can be used to discern the direction and magnitude of changes from the current conditions and continuing trends to develop metrics of success (e.g., see Keane et al. 2009, Safford and Stevens 2017). An evidenced-based approach built on data and the scientific method is the most promising path to promote resilience in forests subject to future wildfires and climate change (Stephens et al. 2016, 2020). Given the historical role of Indigenous land stewardship on many wNA landscapes, combining western science and Indigenous knowledge systems is foundational to intentionally restoring and adapting western forest ecosystems (Kimmerer and Lake 2001, Lake et al. 2017, Roos et al. 2021).

Here, we provide a synthesis of science-based management strategies that include restoring active fire regimes and fostering resilient forest structure and composition. Through a thorough review of the scientific literature, we evaluate the relative effectiveness of forest management strategies. We then address 10 common questions about fuel treatments and forest adaptation to changing climatic and wildfire regimes: (1) Are the effects of fire exclusion overstated? If so, are treatments unwarranted and even counterproductive? (2) Is forest thinning alone sufficient to mitigate wildfire hazard? (3) Can forest thinning and prescribed burning solve the problem? (4) Should active forest management, including forest thinning, be concentrated in the wildland urban interface (WUI)? (5) Can wildfires on their own do the work of fuel treatments? (6) Is the primary objective of fuel reduction treatments to assist in future firefighting response and containment? (7) Do fuel treatments work under extreme fire weather? (8) Is the scale of the problem too great? Can we ever catch up? (9) Will planting more trees mitigate climate change in wNA forests? and (10) Is post-fire management needed or even ecologically justified?

Fuel treatments and active forest management

Biophysical context and socio-cultural considerations.— Much of the literature on adaptive forest management and fuel treatments in wNA pertains to seasonally dry pine and mixed-conifer forests, including ponderosa pine (*Pinus ponderosa*), Jeffrey pine (*P. jeffreyi*), interior Douglas-fir, and mixed-conifer forests of Douglas-fir (*Pseudotsuga menziesii*), grand or white fir (*Abies grandis*, *A. concolor*), and western larch (*Larix occidentalis*) and is concentrated on the western United States. However, as reviewed by Hagmann et al. (*in press*), the effects of fire exclusion are broad reaching and include departures in oak woodlands, mixed broadleaf-conifer forests, and cold forests as well. As we address the topics of forest and fuel management, it is important to provide the context, observation scale, and scope of inference of existing studies to understand where and when active management may be warranted.

Seasonally dry pine and mixed-conifer forests were historically dominated by fire- and drought-tolerant conifers with thick bark; fire-tolerant leaf, branch, and crown morphology; and other adaptations to surviving low- to moderate-intensity surface fires (Agee 1996, Margolis and Malevich 2016, Stevens et al. 2020). Repeated fires removed fuels and created highly varying patterns of individual trees, small tree clumps, and variable sized openings (Jeronimo et al. 2019, Kane et al. 2019). These fuel characteristics collectively contributed to resistance to active crown fires (Ritter et al. 2020) but allowed for individual tree and tree-group torching. Past management and fire exclusion caused tree infilling in many of these forests (Naficy et al. 2016, Hessburg et al. 2019), resulting in substantially denser forests with continuous layered canopies, homogeneous structure, higher density of fire-intolerant species, and high surface fuel loads and fuel ladders connecting surface to crown fuels (Savage et al. 2013, Battaglia et al. 2018, van Mantgem et al. 2018).

Many western oak woodlands and mixed hardwood-pine forests were historically adapted to frequent fire and actively maintained by Indigenous burning practices (Lake et al. 2018). In the absence of frequent fire, oak woodlands and hardwood-conifer forests have been invaded by conifers and other vegetation (Engber et al. 2011, Hoffman et al. 2019). Due to the often extensive fuel ladders and surface fuel loads of contemporary mixed oak-conifer woodlands, reintroducing low-severity fire in forests now dominated by conifers will not likely restore oak woodlands to enable an active fire regime (Barnhart et al. 1996). In some locations, invasion of non-native grasses combined with frequent human ignitions can lead to a decline in oak woodlands and mixed hardwood-pine forests, favoring grassland expansion, and precluding restoration of oak woodlands (Lilley and Vellend 2009).

Moist mixed-conifer and broadleaf deciduous forests (e.g., quaking aspen, black cottonwood, and balsam poplar, *Populus tremuloides*, *P. trichocarpa, and P. balsamifera*) exist throughout wNA, and where they reside in drier climatic settings, they occupy moist sites and valley-bottom locations. These are environments where dense forests with multi-layered canopies are more typical. Historically, moderate- and high-severity fires were common in these topographic settings (Perry et al. 2011,

Hessburg et al. 2019). However, where moist mixed forests were interspersed between dry pine and mixed-conifer forest along topographic and edaphic gradients, low- and moderate-severity fires also commonly occurred (Hagmann et al. 2014, Johnston et al. 2016, Merschel et al. 2018, Ng et al. 2020). Historically, frequent fire favored fire-tolerant tree species and open canopy conditions that were well below carrying capacity of many mixed-conifer forest sites (Hagmann et al. 2021). Indigenous burning also intentionally created patches of meadows, prairies and seasonally dry wetlands in some moist conifer forests (Underwood et al. 2003, Storm and Shebitz 2006). With climate shifting to warmer and drier conditions, managers may reduce the vulnerability of these patches by employing variable density thinning and prescribed fire that favor the likelihood of low- to moderate fire effects rather than high severity by creating tree clumps, gaps, and openings within currently continuous forest canopies (Churchill et al. 2013, Knapp et al. 2017). Where reducing the risk of large patches of high-severity fire is the goal, many of the same strategies used in dry mixed-conifer forests are appropriate to moist mixed-conifer forests (LeFevre et al. 2020). However, small patches of dense and older forest can be embedded within the clumped and gapped tree patterns, and large patches are especially appropriate on north aspects and in valley bottoms (Perry et al. 2011, Hessburg et al. 2015).

Montane cold forests are dominated by thin-barked species such as Engelmann spruce (*Picea engelmanii*), subalpine fir (*A. lasiocarpa*), and lodgepole pine (*P. contorta*), and can include white and black spruce (*P. glauca* and *P. mariana*) further north in the Canadian boreal and subboreal zones (Rowe and Scotter 1973, Agee 1996, Morgan et al. 2008). Departures in these forests are primarily manifested in a loss of burned and recovering patchworks, loss of seral stage and patch size complexity, and high crown fire potential over broad areas (Hessburg et al. 2019, Fig. 1) rather than within-patch changes in tree density and composition. Historical resilience in these forests was largely driven by landscape heterogeneity in the form of patchworks of nonforest vegetation (shrublands, wet and dry meadows) and varied successional and surface fuel conditions, which reduced contagion of dense and layered forests (Stockdale et al. 2019). Indigenous fire stewardship in some cold forests varied post-fire effects to stagger availability of desired resources. The condition of the valued resources (e.g., foods, forage for big game, medicines, basketry materials), fuel loading, and fuel continuity determined the frequency, seasonality, and locations of intentionally burning, where lightning ignitions were too few, or fire effects were insufficient to the maintenance of resources (Lake and Christianson 2019).

Fuel treatments and how they contribute to forest adaptation.—Stephens et al. (2010) recommend four strategies for adapting western forest landscapes to changing climatic and wildfire regimes. They define *resistance* work as that which mitigates expected wildfire effects and protects valued resources, while *realignment* work modifies existing conditions to restore key ecosystem patterns and the processes they drive. Creating *resilient* conditions improves the natural capacity of an ecosystem to respond favorably when unplanned or unanticipated disturbances occur. Finally, they present *response* work as any active facilitation to achieve culturally and ecologically desirable results that are otherwise difficult to achieve. Each of these strategies can play a role in wNA forest management.

As wNA forest ecosystems respond to warmer and drier summers and longer fire seasons, some areas that once supported forests will shift to nonforest (Parks et al. 2019, Coop et al. 2020), and historical fire regimes that resulted from feedbacks between past climate and vegetation may no longer be supported (McWethy et al. 2019). With rapid change and ecological surprises, novel ecosystems and disturbance regimes will emerge, and there is a high level of uncertainty in future ecological outcomes. The combined strategies reviewed in Stephens et al. (2010) can be used to prioritize where adaptive forest management may be the most advisable and effective (Box 1). Furthermore, facilitating ecosystem shifts in portions of the landscape can benefit resilience at landscape and regional scales. For example, certain vegetation types (e.g., shrub and grasslands) may be more adapted to future climate conditions and can contribute to landscape heterogeneity. They also may alter fire behavior patterns towards a reduction in crown fire initiation and spread

There are two main types of management actions to modify forest fuels (termed fuel treatments), and they include (1) reducing surface and canopy fuels via prescribed burning, thinning or other mechanical treatments followed by removal or on-site burning of woody debris, or (2) rearranging fuels including thinning or mechanical treatments without slash reduction. Each type of treatment directs how and where potential energy is stored and released at the scale of forest patches to landscapes, and thresholds to burning.

Fuel reduction.—Common *fuel reduction treatments* include a combination of (1) forest thinning to reduce canopy bulk density and ladder fuels, and (2) prescribed burning or biomass removal to reduce surface fuels, including logging slash from the thinning event and prior fuel accumulations (Reinhardt et al. 2008, Kalies and Yocom Kent 2016). Prescribed burning of logging slash generally includes piling and burning concentrated logging slash and broadcast burning dispersed slash. Forest management projects aimed at fuel reduction in dry or moist mixed-conifer forests and pine, Douglas-fir, or oak woodlands are designed to foster the development of forest structure, composition, and configurations that are more resilient to drought and disturbances. These treatments also commonly reduce

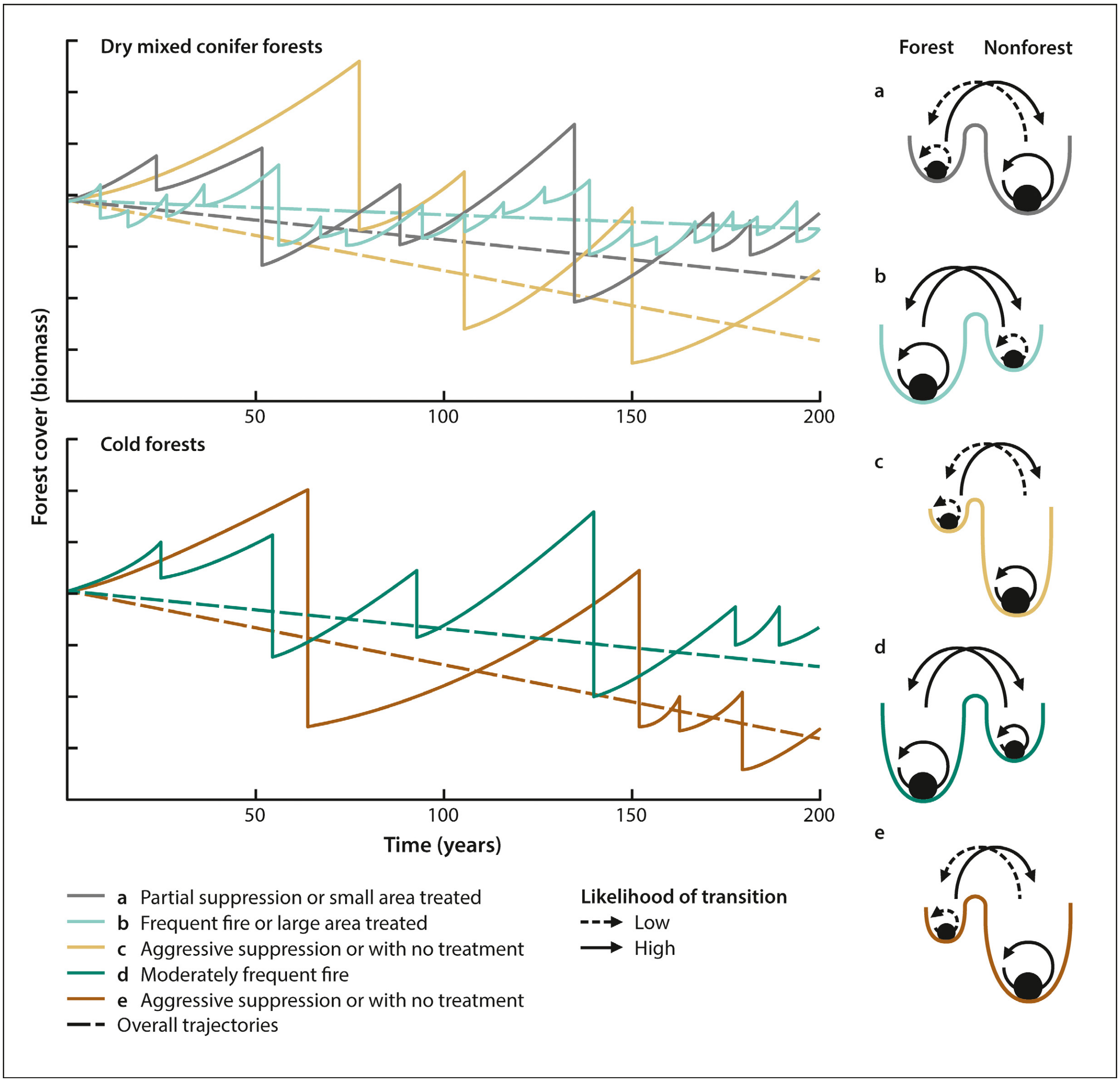


FIG. 1. (A) Dry mixed-conifer forests. Theorized responses of seasonally dry mixed-conifer forest biomass to wildfire and three fire management scenarios under 21st-century climate change. (a) Partial wildfire suppression with only a small fraction of forested landscape treated each year (˜1%). In this scenario, escaped high-severity wildfires are the dominant change agent with a high probability of forest conversion to nonforest as represented in the ball and cup figure by a shallow forest basin of attraction and a deep and broad nonforest basin of attraction. (b) A large percentage of the forested landscape (>50%) is treated either by frequent low and moderate severity fires or fuel reduction treatments with ongoing maintenance. Large wildfires are infrequent, and fire severity within the event perimeter is mostly low and moderate severity as represented in the ball and cup figure by a deep and wide forest basin of attraction and a moderately deep and wide nonforest basin of attraction. (c) Aggressive wildfire suppression with no active fuel reduction treatments; similar to scenario A but with even a higher likelihood of forest to nonforest conversion. (B) Cold forests. Wildfire management scenarios represent two levels of wildland fire management under 21st-century climate change. (d) Cold forest area treated with moderately frequent fires of moderate and high severity. Because large fire events are relatively rare, forest regeneration is supported by patchworks of remnant forest, represented by a deep and wide forest basin of attraction. (e) Aggressive fire suppression with no active fuel treatments. In this scenario, escaped wildfires are the major change agent through large, mostly high severity fires. Forest regeneration is limited by large, high severity fire events, and conversion to nonforest is common; represented by a shallow and narrow forest basin of attraction and a deep and broad nonforest basin of attraction.

surface fuel loads to promote lower flame lengths, surface fire intensity and spread, and a reduction in crown fire potential (Agee and Skinner 2005). Forest thinning in these forest types is aimed at retaining larger, more fire-resilient tree species, and restoring open canopy structure. For example, the individuals, clumps, and openings (ICO) method selects trees and tree groups to impart spatial heterogeneity to the forest by varying the

Box 1. Defining restorative and adaptive management

Ecosystem restoration is actively assisting the recovery of an ecosystem that has been degraded, damaged, or transformed (Holl 2020). Adaptive management is a learning-by-doing method of responding to ecosystem changes, informed by effectiveness monitoring (Lyons et al. 2008, Larson et al. 2013*b*). Recent reviews examine in detail research on adaptive and restorative forest fuel treatments, including mechanical thinning, prescribed and Indigenous cultural burning, and management of unplanned ignitions, and their relative effectiveness at mitigating future wildfire spread and severity (Fulé et al. 2012, Stephens et al. 2012, Martinson and Omi 2013, Ryan et al 2013, Kalies and Yocom Kent 2016). Across seasonally dry forests, a promising finding is that treatments involving prescribed or cultural burning or effectively managed wildfires generally mitigate the spread and severity of subsequent wildfires for a period of time after treatment (5–20 yr, depending on site productivity, vegetation, and climate), and are often more effective than mechanical treatments without follow-up prescribed burning (Prichard et al. 2017). Use of these management techniques can therefore improve forest resilience and resistance to change under a warmer, drier climate.

Treatments designed to restore or adapt fire-excluded forests to a changing climate must foster ecosystem resilience and conserve native biodiversity. For example, restoration treatments are often designed to enhance plant vigor, favor fire-adapted species, and create open forest structures, all with the objective of increasing resilience and resistance to climatic warming and severe wildfires (Lehmkuhl et al. 2007, Reinhardt et al. 2008, North et al. 2012). An added benefit of most restorative treatments is that wildland fuel hazard is also reduced (Fulé et al. 2001, Brown et al. 2004). Fire-less fuel reduction treatments rarely mimic the broad role of fire (Reinhardt et al. 2008), which performs many cultural and ecological functions, e.g., nutrient cycling, facilitating tree regeneration by exposing mineral soils, promoting valued cultural and aesthetic resources (Marks-Block et al. 2019). As a result, any area treated using mechanical fuel treatments alone rarely restores fire-adapted ecosystems.

distribution of forest and non-forest cover to achieve a low edge to interior ratio with the goals of refostering drought tolerance and reducing the probability of crown fire (Larson and Churchill 2012, Churchill et al. 2017). Recent evidence suggests that low-intensity fire alone may not increase resilience because it is not sufficiently lethal to shade-tolerant species that established during an extended period of fire exclusion (e.g., Douglas-fir, grand fir, white fir, incense-cedar [*Calocedrus decurrens*]; Cocking et al. 2014, Huffman et al. 2018, Eisenberg et al. 2019). Methods such as ICO are intended to emulate the structural patterns maintained by frequent fires and can be employed where single entry fires may not achieve restoration goals.

Due to altered stand conditions, restoring an active fire regime and reducing climate vulnerability often requires either a managed wildfire that significantly thins forests, consumes fuels, and favors fire-resistant, larger trees (Holden et al. 2010, Kane et al. 2015), or coupled mechanical thinning and prescribed or cultural burning treatment followed by regular maintenance burning (Stephens et al. 2012). Unplanned wildfires that consume surface fuels can also be considered fuel reduction treatments under moderate fire weather conditions (North et al. 2012, Prichard et al. 2017). Mechanical treatments that involve thinning and off-site biomass transport can also be effective fuel reduction surrogates where infrastructure and economics allow (North et al. 2015*a*). In all cases, fuel reduction treatments can be effective at mitigating subsequent wildfire behavior and effects for a period of time after treatment until surface and canopy fuels accumulate through vegetation growth and deposition (Keane et al. 2015).

The key to effective fuel reduction is that it creates gaps in surface and canopy fuel structures and reduces the potential for contagious crown fire initiation and spread (Reinhardt et al. 2008, Martinson and Omi 2013, Fig. 2A). Depending upon the scale of a wildfire event and the underlying climate and weather conditions, past fuel reduction treatments can mitigate fire spread and intensity at very fine to coarse spatial scales (Fulé et al. 2012, Prichard et al. 2017). For example, in a fire-maintained pine forest or savanna, frequent understory burning can maintain low loads of pine needle duff and litter, fine wood and grass to support low-intensity surface fires. In these forest types, the threshold for high-severity fire is only crossed during extreme fire weather and fire behavior, often involving plume-driven fire spread from adjacent forests (Agee and Skinner 2005, Lydersen et al. 2014).

Fuel rearrangement.—Without associated reduction of surface fuels, mechanical thinning and mastication treatments are examples of *fuel rearrangement* treatments (Fig. 2B). Commercial or pre-commercial forest thinning reduces the continuity of tree crowns, their bulk density, and their propensity for spreading crown fire. Consequently, thinning without prescribed burning is considered both a *reduction* of canopy and ladder fuels and a *rearrangement* of fuels from the canopy to the forest floor (Pollet and Omi 2002). Where canopy thinning results in augmented surface fuels, fire behavior and

FIG. 2. Representative photos of (A) fuel reduction treatment (maintenance surface fire in a previously thinned and burned forest); (B) fuel rearrangement (forest residues following mechanical thinning); and (C) fuel accumulation (fire excluded forest with grand fir infilling around western larch trees). Photo credits: Roger Ottmar, Susan Prichard, and John Marshall.

severity can be amplified rather than diminished (Safford et al. 2009, Prichard et al. 2010). Furthermore, many fire-excluded forests have elevated surface fuels associated with more than a century of fire exclusion (Knapp et al. 2013, Keane et al. 2015). Effective treatment therefore necessitates prescribed burning that is intense enough to reduce surface and ladder fuels such that the likelihood of a subsequent intense fire is reduced (Stephens et al. 2012). Wildfires that result in substantial tree mortality may offer a short-term fuel reduction, but over longer time periods (15–25 yr), downed wood accumulations from snag and branch fall can elevate surface fuels and create conditions for high-intensity reburn events (Stevens-Rumann et al. 2012, Dunn and Bailey 2016, Johnson et al. 2020). As such, moderate to high-severity wildfires are generally considered a type of longer-term fuel rearrangement (Lydersen et al. 2019*a*).

Development of landscape mosaics.—Intentional management of landscapes involves the broad-scale planning and spatial design of treatments, including determining where they are most effective on the landscape and assessing how individual treatments will interact with fire over space and time (Ager et al. 2010, Falk et al. 2019). Many historical landscapes, influenced by lightning and Indigenous ignitions, supported a hierarchical patchwork of forest and nonforest vegetation at coarse spatial scales in addition to meso- and fine-grained heterogeneity of forest age classes and vulnerability to fire (Hessburg et al. 2019, Hagmann et al. 2021). Managed landscape mosaics can be designed to restore more characteristic patchworks of open and closed canopy vegetation of different patch sizes, tree ages, and forest densities, and of fuel contagion to facilitate restoring fire as a dynamic and beneficial ecological process (Hessburg et al. 2015).

Fuel treatments that modify within-stand structure to remove small trees and reduce surface fuels while retaining large, more fire-resistant trees and variable stand structure (Stephens et al. 2021) are most appropriate in dry pine, dry to moist mixed-conifer forests and oak woodlands, particularly where there is evidence that older fire-resistant species have been or are being replaced by younger fire-sensitive species (e.g., Yocom-Kent et al. 2015). This mirrors the fine- to meso-scale (i.e., 1–10,000 ha) heterogeneity in forest structure that characterized these frequent-fire forest types historically (Hessburg et al. 2019, Hagmann et al. 2021). In cold forests characterized by greater landscape-scale heterogeneity, fuel treatments including managing unplanned wildfires may be more appropriate at larger scales, particularly where landscape-scale heterogeneity has been lost (Hessburg et al. 2019, Hagmann et al. 2021).

Within this context, reserves and other no-treatment areas can be designated where fuels are left to accumulate over time (Fig. 2C). Competing resource management objectives and consideration of values at risk often inevitably lead to management areas where fuel reduction treatments are not allowed and wildfires are actively suppressed. Examples include late-successional reserves, riparian reserves, and other locations where wildland fires and fuel reduction treatments are restricted to facilitate habitat development. Over time, surface and canopy fuel accumulations and wildfire dynamics will threaten the objectives of these reserved areas (Van de Water and North 2011, Reilly et al. 2018). Stationary reserves will be difficult to maintain in areas where wildfires are the disturbance engine that drives the ecosystem.

TEN COMMON QUESTIONS ABOUT ADAPTIVE FOREST MANAGEMENT

Although the need to increase the pace and scale of fuel treatments is broadly discussed in scientific and policy arenas (Franklin and Johnson 2012, North et al. 2012, Kolden 2019), there is still confusion and disagreement about the appropriateness of forest and fuel treatments. For example, recent publications have questioned whether large, high-severity fires are outside of the historical range of variability for seasonally dry forests, and whether the risk of high-severity fire warrants large-scale treatment of fire-prone forests (Bradley et al. 2016, DellaSala et al. 2017). Others have questioned whether

intentional management, including forest thinning, is effective or justified outside of the wildland urban interface (Moritz et al. 2014, Schoennagel et al. 2017). Furthermore, debates around the management of fire-adapted forests are occurring within the context of long running conflicts over timber production on public lands, especially federal lands, leading to questions about science-based benefits of management treatments where they align with economic incentives (Daniels and Walker 1995). Currently, management strategies employing active fire suppression and limited use of fuel reduction treatments are common for most public land management agencies.

Among the many challenges to proactive management on public lands (e.g., funding, adequate and qualified personnel, smoke impacts, and weather and fuel conditions that fall within burn prescription parameters), uncertainty in the scientific literature about forest management and fuel treatments is commonly cited in planning process-public comment periods (Spies et al. 2018, Miller et al. 2020). In the following sections, we examine 10 common questions about forest management and fuel treatments. We summarize them in Table 1 and provide key citations that examine these questions. For each topic, we evaluate the strength of evidence in the existing scientific literature concerning each topic. Our goal is to help managers, policy makers, informed public stakeholders, and others working in this arena to establish a robust scientific framework that will lead to more effective discussions and decision-making processes, and better outcomes on the ground. Additional citations for each question are listed in Appendix S1.

Are the effects of fire exclusion overstated? If so, are treatments unwarranted and even counterproductive?

Concerns about forest thinning and other forms of active management are sometimes based on the assumption that contemporary conditions and fire regimes in dry pine and mixed-conifer forests are not substantially departed from those maintained by uninterrupted fire regimes (Hagmann et al. 2021). This perspective does not accurately reflect the breadth and depth of scientific evidence documenting the influence of over a century of fire exclusion. Support for the suggestion that ecological departures associated with fire exclusion are overestimated has repeatedly failed independent validation by multiple research groups (Hagmann et al. 2021). In addition, these arguments fail to consider widespread Indigenous fire uses that affected landscape scale vegetation conditions linked to valued cultural resources and services, food security, and vulnerability to wildfires (Lake et al. 2018, Power et al. 2018). As is explored in the following sections, a number of forest management and treatment strategies are shown to be highly effective. Site conditions and history are always important considerations. Moreover, there is no one-treatment-fits-all approach to forest adaptation.

Evidence from a broad range of disciplines documents widespread, multi-regional 20th-century fire exclusion in interior forested landscapes of wNA (see a detailed reference list and discussion in Hagmann et al. 2021). Collectively, these studies reveal extensive changes in tree density, species and age composition, forest structure, and continuity of canopy and surface fuels. Forests that were once characterized by shifting patchworks of forest and nonforest vegetation (i.e., grasslands, woodlands, and shrublands) in the early 20th-century gradually became more continuously covered in forest and densely stocked with fuels (Fig. 4).

However, for over two decades, a small fraction of the scientific literature has cast doubt on the inferences made from fire-scar based reconstructions and broader landscape-level assessments to suggest that estimates of low- to moderate-severity fire regimes from these studies are overstated. Hagmann et al. (*in press*) examine this counter-evidence in detail and identify critical flaws in reasoning and methodologies in original papers and subsequent re-application of these methods in numerous geographic areas. Subsequent research shows that studies relying on Williams and Baker (2011) methods for estimating historical tree densities and fire regimes overestimate tree densities and fire severity (see also Levine et al. 2017). Moreover, established tree-ring fire-scar methods more accurately reconstruct known fire occurrence and extent. Other studies, also based on the methods of Williams and Baker (2011), conflate reconstructed low-severity, high-frequency fire regimes with landscape homogeneity. These interpretations disregard critical ecosystem functions that were historically associated with uneven-aged forests embedded in multi-level fine-, meso- and broad-scale landscapes. By extension, claims that low-severity fire regimes are overestimated then imply that large, high-severity fires were a regular occurrence prior to the era of European colonization. Such interpretations may lead to the conclusion that recent increases in high-severity fire are still within the historical range of variability, and that there is no need of restorative or adaptive treatments (Hanson and Odion 2014, Odion et al. 2014, Baker and Hanson 2017).

Indeed, research from across wNA has shown that high-severity fire was a component of historical fire regimes, and that fires of all severities are currently in deficit (Parks et al. 2015*b*, Reilly et al. 2017, Haugo et al. 2019, but see Mallek et al. 2013). However, reanalysis of the methods of Baker and others shows that their methods inherently overestimate fire severity and the frequency and area affected by high-severity fire (Fulé et al. 2014, Hagmann et al. 2021). In addition, high-severity patches in recent fires are less heterogeneous and more extensive than the historical range of variability for forests characterized by low- and moderate-severity fire regimes (Stevens et al. 2017, Hagmann et al. 2021). Finally, research across wNA reveals key climate-vegetation-wildfire linkages,

Table 1. Ten common questions about active forest management.

Question	Summary of evidence	Key citations
(1) Are the effects of fire exclusion overstated? If so, are treatments unwarranted and even counterproductive?	Broad-scale evidence of fire exclusion is strong across disciplines and western forest ecosystems. Although high severity fire was a component of many historical fire regimes, the frequency and extent of high severity fire over the past few decades is outside the range of historical range of variability	Hessburg et al. (2005), Reynolds et al. (2013), Stine et al. (2014), Safford and Stevens (2017), Stephens et al. (2020), Hagmann et al. (2021)
(2) Is forest thinning alone sufficient to mitigate wildfire hazard?	Thinning alone can sometimes mitigate fire severity, but through residual logging slash, desiccation of understory fuels, and increased surface wind flow without accompanying surface fuel reduction, thinning can contribute to high-intensity surface fires and abundant mortality	Stephens et al. (2009), Fulé et al. (2012), Martinson and Omi (2013), Kalies and Yocom Kent (2016)
(3) Can forest thinning and prescribed burning solve the problem?	Although thinning and prescribed burning have been shown to be highly effective, not all forests are appropriate for this treatment (e.g., thin-barked species common in cold mixed-conifer forests). This type of fuel treatment is also not appropriate for wilderness and other roadless areas	DellaSala et al. (2004), Battaglia and Shepperd (2007), Reinhardt et al. (2008)
(4) Should active forest management, including forest thinning, be concentrated in the wildland urban interface (WUI)?	The majority of designated WUI is in private ownership and hence these lands are sometimes more difficult to treat than public lands. Treating dry and moist mixed-conifer forests beyond WUI buffers can modify fire behavior and change the intensity of wildfires arriving at communities	Kolden and Brown (2010), Bladon (2018), Hallema et al. (2018), Kolden and Henson (2019), Schultz et al. (2019)
(5) Can wildfires on their own do the work of fuel treatments?	Unplanned fires that escape suppression often burn under extreme fire weather and can have severe wildfire effects. In contrast, prescribed burns and managed wildfires generally burn under more moderate weather conditions and contribute to variable fire effects and surface fuel reduction that can mitigate future wildfire severity	Miller and Safford (2012), Parks et al. (2015*a*, 2016), Prichard et al. (2017), Stevens et al. (2017), Kane et al. (2019), Huffman et al. (2020), Rodman et al. (2020)
(6) Is the primary objective of fuel reduction treatments to assist in future firefighting response and containment?	Although fuel reduction treatments can assist in suppression operations, primarily using fuel treatments to suppress future wildfires actually contributes to wildland fire deficit. Adaptive treatments in fire-adapted landscapes aim to restore the patch to landscape role of fire as an ecological process, reduce fire effects and need for aggressive suppression when the fire next occurs	Reinhardt et al. (2008), Safford et al. (2012), Stephens et al. (2020)
(7) Do fuel treatments work under extreme fire weather?	Fire behavior associated with persistent drought, high winds and column-driven spread are associated with higher burn severity in western North American forests. However, strong scientific evidence across dry and moist mixed conifer forests demonstrates effectiveness at mitigating burn severity, often even under extreme fire weather conditions	Arkle et al. (2012), Yocom-Kent et al. (2015), Povak et al. (2020), Prichard et al. (2020)
(8) Is the scale of the problem too great? Can we ever catch up?	The current pace and scale of treatments is decidedly inadequate to restore fire-resilient and climate adapted landscapes. However, evidence strongly supports that expanded use of fuel reduction treatments can be effective	Collins et al. (2009), North et al. (2012), Parks et al. (2015*a*, 2016), Ager et al. (2016), Barros et al. (2018), Liang et al. (2018)
(9) Will planting more trees mitigate climate change in wNA forests?	Temperate rainforests and other wet forests have the capacity to store and sequester high amounts of forest carbon. However, planting to increase tree density and continuity in fire-prone forests is unsustainable due to high fire danger, anticipated climatic water deficits and drought stress	Thompson et al. (2007), Veldman et al. (2019), Holl and Brancalion (2020)
(10) Is post-fire management needed or even ecologically justified?	Active forest and fuels management may be required beyond the initial fire response in order to promote future forest resilience to disturbance and climate change. Due to fire exclusion, uncharacteristically dense patches of dead trees may contribute to high-severity reburns as they fall and create heavy surface fuel accumulations	Peterson et al. (2015), Lydersen et al. (2019*a*), North et al. (2019)

Note: Western North America is abbreviated wNA.

where fire frequency, extent, and severity all increase with increasing climatic warming, suggesting that observed trends in fire patterns are commensurate with predicted relationships with ongoing climate change (McKenzie and Littell 2017, Parks and Abatzoglou 2020).

Another perspective on this debate contends that whether historical records can be agreed upon is of ancillary importance. Adaptive forest management and fuel reduction treatments are primarily aimed at increasing forest resilience and/or resistance to climate change, fire and other disturbances, which has positive societal and ecological impacts that do not require justification based on historical conditions, particularly given the no-analog present and future that climate change presents (Freeman et al. 2017). For example, the most concerning contemporary high-severity fire events are associated with large patches of complete stand replacement (Miller and Quayle 2015, Lydersen et al. 2016). In some cases, high-severity fire events convert forests to shrubland and grassland assemblages as alternative stable states in uncharacteristically large patches (Falk et al. 2019, Kemp et al. 2019, Stevens-Rumann and Morgan 2019). As such, a critical forest management concern is that high-severity wildfires are accelerating rates of vegetation change, forest conversion, and vulnerability of native habitats in response to a warming climate.

Is forest thinning alone sufficient to mitigate wildfire hazard?

While "thin the forest to reduce wildfire threat" is commonly cited in the popular media, the capacity for thinning alone to mitigate wildfire hazard and severity is not well supported in the scientific literature. Thinning treatments require strategic selection of trees to target fuel ladders and fire-susceptible trees, along with a subsequent fuel reduction treatment (Jain et al. 2020). When thinning is conducted without accompanied surface fuel reduction, short and long-term goals may not be realized.

Thinning from below reduces ladder fuels and canopy bulk density concurrently, which can reduce the potential for both passive and active crown fire behavior (Agee and Skinner 2005). For instance, Harrod et al. (2009) found that thinning treatments that reduced tree density and canopy bulk density and increased canopy base height significantly reduced stand susceptibility to crown fire compared to untreated controls. Furthermore, large-diameter trees and snags that provide essential wildlife habitat and other ecosystem values can be retained and fuels can be deliberately removed around these structures using this approach (Lehmkuhl et al. 2015). Where wood from treatments can be marketed, revenues from thinning help to sustain broader management goals on public lands. For example, some landscape restoration collaboratives seek to reinvest profits from commercially viable thinning to off-set costs associated with more labor-intensive manual thinning and prescribed or cultural burning needs (Schultz and Jedd 2012).

Some studies show that thinning alone can mitigate wildfire severity (e.g., Pollet and Omi 2002, Prichard and Kennedy 2014, Prichard et al. 2020), but across a wide range of sites, thin and prescribed burn treatments are most effective at reducing fire severity (see reviews by Fulé et al. 2012, Martinson and Omi 2013, Kalies and Yocom Kent 2016). On most sites, thinning alone achieves a reduction of canopy fuels but contributes to higher surface fuel loads. If burned in a wildfire, these fuels can contribute to high-intensity surface fires and elevated levels of associated tree mortality (e.g., Stephens et al. 2009, Prichard and Kennedy 2012). When trees are felled and limbed, fine fuels from tree tops and branches (termed activity fuels) are re-distributed over the treatment area, thereby increasing surface fuel loads (Martinson and Omi 2013). Mechanical fuel reduction treatments of these activity fuels are possible, but in many locations, biomass removal and utilization (e.g., for bioenergy) after thinning treatments can be cost-prohibitive due to long hauling distances and the economic and technological challenges of building new biomass facilities (Hartsough et al. 2008). Mastication equipment is sometimes used to shred understory trees and shrubs into smaller woody fragments, which are then redistributed and left on site (Kane et al. 2009). However, following mastication, surface fuels are temporarily elevated, and masticated stands that burn in wildland fires can cause deep soil heating from long-duration smoldering combustion and elevated fire intensities (Kreye et al. 2014).

Other unintended consequences of thinning without concomitant reduction in surface fuels can occur. For instance, decreasing canopy bulk density can change site climatic conditions (Agee and Skinner 2005). Wildfire ignition potential is largely driven by fuel moisture, which can decrease on drier sites when canopy bulk density is reduced through commercial thinning (e.g., Reinhardt et al. 2006). Reduced canopy bulk density can lead to increased surface wind speed and fuel heating, which allows for increased rates of fire spread in thinned forests (Pimont et al. 2009, Parsons et al. 2018). Other studies show no effect of thinning on surface fuel moisture (Bigelow and North 2012, Estes et al. 2012), suggesting that thinning effects on surface winds and fuel moisture are complex, site specific, and likely vary across ecoregions and seasons.

In summary, although the efficacy of thinning alone as a fuel reduction treatment is questionable and site dependent, there exists widespread agreement that *combined* effects of thinning plus prescribed burning consistently reduces the potential for severe wildfire across a broad range of forest types and conditions (Fig. 3; Fulé et al. 2012, Kalies and Yocom Kent 2016, Stephens et al. 2021). Given this broad consensus in the scientific literature, some authors suggest that forest thinning should be considered in the context of wildfire hazard abatement, ecological restoration and adaptation, and revitalization of cultural burning (Lehmkuhl et al. 2007, Hessburg et al. 2015, Huffman et al. 2020). Where restoring resilient forest composition and structure and reducing future wildfire hazard are goals of management (Koontz et al. 2020), combined thinning and burning approaches will provide ecological and wildfire-risk reduction benefits (Knapp et al. 2017).

FIG. 3. Active forest restoration treatment, Sinlahekin Wildlife Refuge, Washington Department of Fish and Wildlife. Top left: multi-layered, dense dry mixed conifer forest after 100 yr of fire exclusion. Top right: residual forest after a variable density thinning treatment. Bottom right: treated condition after pile and broadcast burning. Bottom left: post-wildfire photo after the 2015 Lime Belt fire. *Photo credit: John Marshall.*

Can forest thinning and prescribed burning solve the problem?

Fire has been a tool that has been actively used for millennia. Indigenous burning practices maintained prairies, oak and pine savannas, riparian areas, mixed-conifer, hardwood, and dry forests, and high mountain huckleberry and beargrass assemblages for food, medicine, basketry and other resources (Trauernicht et al. 2015, Roos et al. 2021). Following prolonged fire exclusion, many seasonally dry forest landscapes that were once frequently burned now are densely stocked with multi-layered canopies that often require thinning prior to restoring fire (North et al. 2012, Ryan et al. 2013). Prescribed burning on its own and in combination with mechanical thinning are essential fuel reduction treatments with demonstrated effectiveness in reducing fire severity, crown and bole scorch, and tree mortality compared to untreated forests (Safford et al., 2012*a*,*b*, Kalies and Yocom Kent 2016). Thinning and burning in partnership with local Indigenous knowledge and practice can support culturally valued practices, traditions, livelihoods, and food and medicine security (Sowerwine et al. 2019).

Although the use of prescribed burning, often in combination with mechanical thinning, has been shown to be highly effective at mitigating wildfire severity and increasing forest resilience to drought, insects and disease (Hood et al. 2015), these treatments alone cannot address forest management challenges across wNA. Fuel reduction treatments are not appropriate for all conditions or forest types (DellaSala et al. 2004, Reinhardt et al. 2008, Naficy et al. 2016). In some mesic forests, for instance, mechanical treatments may increase the risk of fire by increasing sunlight exposure to the forest floor, drying surface fuels, promoting understory growth, and increasing wind speeds that leave residual trees vulnerable to wind throw (Zald and Dunn 2018, Hanan et al. 2020). Furthermore, prescribed surface fire is difficult to implement in many current mesic forests since fire readily spreads into tree crowns via abundant fuel ladders and can result in crown fires. In other forest types such as subalpine, subboreal, and boreal forests, low crown base heights, thin bark, and heavy duff and litter loads make trees vulnerable to fire at any intensity (Agee 1996, Stevens et al. 2020). Fire regimes in these forests, along with lodgepole pine, are dominated by moderate- and high-severity fires, and applications of forest thinning and prescribed underburning are generally inappropriate. However, landscape burning and maintenance of high elevation forests and meadows is part of cultural burning, and high-intensity crown fire is used

Table 2. Examples of wildfire management of unplanned ignitions and the influence of past wildfires in national parks and wilderness areas.

Area	Management objective	Study findings	Biophysical setting	Reference
North Rim Grand Canyon National Park, AZ	Restoring fire; created strategic fuel reductions to allow for natural fire to return	Fires have thinning effect on small diameter trees along with fine fuel and coarse wood consumption	dry ponderosa pine forest and shrublands; cold dry mixed conifer forests	Fulé and Laughlin (2007), Stoddard et al. (2020)
Saguaro Wilderness, AZ	Sky islands; 30 yr of repeated wildland fires	Repeat fires have reduced small density trees but medium trees are still denser than historical stand structures probably supported	dry ponderosa pine forest and shrublands	Holden et al. (2007), Hunter et al. (2014)
Hualapai tribal lands, AZ	Compared fire scars with modern use of low-intensity prescribed burning	Prescribed fires since the 1960s approximate the frequent surface fires of historical record but could incorporate greater variability in temporal schedules of burning	Dry ponderosa pine forests	Stan et al. (2014)
Gila/Aldo Leopold Wilderness, NM	Restore fire as natural process Surface loads and continuity drive high fire frequency on productive sites	Low severity fires beget low severity fires, and high severity fires tend to reburn at high severity in flammable shrub fields. Previous fires reduce size of subsequent fires for a short period of time	dry ponderosa pine forest and shrublands; dry mixed conifer forest; some cold forest	Rollins et al. (2002), Holden et al. (2007, 2010), Hunter et al. (2014), Parks et al. (2014, 2015*a*, 2016, 2018), Holsinger et al. (2016)
Zion National Park, UT	Science-based fire management plan including managed wildfires, prescribed burning, and hazardous fuel reduction	Repeat prescribed fires reduce probability of crown fire and increased grass and forb cover, but not tree density or shrub cover	dry ponderosa pine forest and shrublands	Brown et al. (2019)
Yosemite National Park (YNP), CA	Restore fire as natural process; began with fires within the park interior and gradually worked outward to allow for more fires throughout park	High severity burns favor flammable shrub fields, which perpetuate high severity reburns. Low severity burns perpetuate low severity burns		Boisramé et al. (2017), Collins et al. (2009), Coppoletta et al. (2016), Scholl and Taylor (2010), Thode et al. (2011), van Wagtendonk et al. (2012)
Sequoia and Kings Canyon National Parks, Giant Sequoia National Monuments, CA	Restore fire as natural process	In red fir forests, repeated low- to moderate-severity fire can restore structural heterogeneity		Meyer et al. (2015)
Frank Church – River of No Return Wilderness, ID	Restore fire as natural process	Burn severity is lower within recent fire areas and increases with time since fire. Previous fires reduce size of subsequent fires	dry mixed conifer forests and cold forests	Teske et al. (2012), Parks et al. (2014, 2015*a*, 2016, 2018), Holsinger et al. (2016)
Bob Marshall Wilderness Area, MT	Restore fire as natural process	Previous fires reduce size of subsequent fires	cold mixed conifer forests, Rocky Mountains	Belote et al. (2015), Holsinger et al. (2016), Keane et al. (2006), Larson et al. (2013*a*), Parks et al. (2015*a*, 2016, 2018), Teske et al. (2012)
Selway-Bitterroot Wilderness Complex, ID and MT	Restore fire as natural process; moisture content of large fuels and tree crowns drive fire frequency (higher on drier sites)	Previous fires reduce size of subsequent fires	cold mixed conifer and subalpine forests	Rollins et al. (2002), Parks et al. (2015*a*), 2016, 2018), Barnett et al. (2016*a*), Holsinger et al. (2016), Morgan et al.

Table 2. Continued.

Area	Management objective	Study findings	Biophysical setting	Reference
				(2017), Teske et al. (2012)
Banff, Kootenay and Yoho National Parks (NP), BC & Alberta, Canada	Guard fires to allow for more natural ignitions to burn within park; restoration of aspen and grasslands (bison habitat)	Multiple prescribed burns to reduce dense lodgepole pine (LPP) and allow aspen to regenerate	cold mixed conifer and subboreal forests, Rocky Mountains	White (1985), Park et al. (2019)
Wood Buffalo National Park, AB and NWT, Canada	Restore and maintain fire as natural process	Fire severity is influenced by pre-fire stand structure and composition, topoedaphic context, and fire weather at time of burning. Burned areas less likely to burn again for 33 yr, though this decreases in drought years	vegetation is representative of the western Canadian boreal forest	Parks et al. (2018), Thompson et al. (2017), Whitman et al. (2019)

Note: State and province abbreviations are AZ, Arizona; NM, New Mexico; ID, Idaho; MT, Montana; BC, British Columbia; AB, Alberta; NWT, North West Territory.

operationally on national forests and parks within the United States and Canada for landscape restoration objectives (Table 2).

Even where socially and ecologically appropriate, thinning and low-intensity prescribed burning generally require repeated treatments to meet fuel reduction objectives. For example, without prior thinning, low-intensity prescribed fire, on its own, may not consume enough fuel or cause enough tree mortality to change forest structure and reduce crown fire hazard (e.g., Lydersen et al. 2019*b*). In contrast, prescribed burns in heavy slash may result in high tree mortality. The first harvest entry into fire-excluded stands often leaves high surface fuel loads and dense understories that require one or more prescribed burning treatments to reduce surface and ladder fuels (Goodwin et al. 2018, Korb et al. 2020). Thus, it often takes multiple treatments and/or fire entries, as well as ongoing maintenance, to realize resilience and adaptation goals (Agee and Skinner 2005, Stevens et al. 2014, Goodwin et al. 2020). Given the extent and variability of forest ecosystems that have experienced prolonged fire exclusion, active forest management can be only one tool to increase adaptation to climate and future fires.

Although thinning and prescribed burning have been shown to be highly effective, the current scale and pace of these treatments do not match the scale of the management challenge (Barnett et al. 2016*b*, Kolden 2019). Mechanical treatments are constrained by land management allocations and their enabling legislation (e.g., wilderness and roadless areas), operational constraints (e.g., steep slopes, distance to roads, costs), and administrative boundaries (e.g., riparian areas, areas managed for species of concern). In the central Sierra Nevada for example, these constraints, combined with large areas of non-productive timberland that are unsuitable for commercial treatment due to steep slopes or distance from roads, left only 28% of the landscape available for mechanical thinning and prescribed burning treatments (North et al. 2015*a*). In the remaining area, prescribed burning alone and/or use of managed wildfires may be suitable replacement treatments (Boisramé et al. 2017, Barros et al. 2018). However, prescribed fire-only treatments are frequently limited by cost, liability, air quality regulations, equipment availability, personnel capacity and training, and the need for ongoing maintenance treatments (Quinn-Davidson and Varner 2012, Schultz et al. 2019).

In light of these constraints, some researchers and managers have called for the expanded use of landscape-scale prescribed burns and managed wildfires in addition to fuel reduction treatments as a promising approach to expand the pace and scale of adaptive management (Question 5). Increasingly collaborative restoration partnerships with Indigenous cultures can increase opportunities for re-instating tribal stewardship practices (Lake et al. 2018, Long and Lake 2018). Under appropriate weather and safety conditions, and where infrastructure is not at risk, managed wildfire may serve as a useful and cost-effective tool for reintroducing wildfire to fire-excluded forests and achieve broad-scale management goals.

Should active forest management, including forest thinning, be concentrated in the wildland urban interface (WUI)?

A question often asked by land managers is where to locate fuel treatments to maximize their advantage while minimizing adverse impacts. The 2000 National Fire Plan

(USDA and USDI 2001) and the 2002 Healthy Forests Initiative identified the need to reduce wildfire risk to people, communities, and natural resources. The 2003 Healthy Forests Restoration Act (HFRA, Congress.gov, 2020) then specified that >50% of fuel reduction funding be spent on projects within the Wildland Urban Interface (WUI), and it reduced environmental review within 2.41 km (1.5 miles) of at-risk communities. The significant increase in homes lost and suppression dollars spent in the WUI in subsequent years (Mell et al. 2010) has catalyzed extensive research on the WUI environment and population expansion into wildlands (Radeloff et al. 2018). Subsequent studies demonstrating fuel treatment effectiveness in the WUI (Safford et al. 2009, Kennedy and Johnson 2014) and spatial methods for optimizing WUI fuel treatments (Bar Massada et al. 2011, Syphard et al. 2012) could be taken to suggest that most fuel reduction should be implemented in the WUI to protect homes and lives.

However, prioritizing the WUI-only for fuel reduction treatments is often too narrow in scope to address broader landscape-scale objectives. For example, Schoennagel et al. (2009) found that more than two-thirds of the area within a 2.5 km radius of at-risk communities was privately owned and unavailable for federally funded fuel treatments. This finding partly elucidates why most hazard reduction fuel treatments are implemented outside of HFRA designation. Fuel treatments on federal lands near communities may also be significantly more difficult, expensive, and risky to implement, while air quality regulations and associated risks create disincentives to treating near homes. Alternatively, agencies may be able to meet both annual prescribed burning accomplishment targets and ecological objectives in areas more distant from the WUI with fewer risks, less money, and fewer personnel (Kolden and Brown 2010, Schultz et al. 2019). Further, there is increasing evidence that treating fuels across larger spatial extents in strategically planned wildland locations, rather than immediately adjacent to WUI, can indirectly reduce risk to communities (Smith et al. 2016, Bowman et al. 2020). Benefits of this strategy include increased initial attack and short-term suppression effectiveness, reduced crown fire potential and ember production, reduced smoke impacts to communities, and increased forest resilience (Ager et al. 2010, Stevens et al. 2016).

Fuel reduction treatments also can support cultural, ecological, ecosystem service, and management objectives beyond the WUI. For example, treatments that restore the ecological resilience of old-growth forests and patches with large and old trees are critical to long term maintenance of wildlife habitats (Hessburg et al. 2020) of seasonally dry forests and terrestrial carbon stocks, and slowing the feedback cycle between fire and climate change (Hurteau and North 2009). Treatments in watersheds that are distant from the WUI and protect municipal and agricultural water supplies are critical to minimizing high-severity fire impacts that can jeopardize clean water delivery (Bladon 2018, Hallema et al. 2018). For example, post-fire erosion and debris flows may cause more detrimental and longer-term impacts to watersheds than the wildfires themselves (Jones et al. 2018, Kolden and Henson 2019).

Finally, treated areas outside the WUI can serve as defensible positions for fire suppression personnel that can be used to establish control lines or allow for more flexible suppression strategies, freeing up resources to protect WUI infrastructure or forests in another area (Thompson et al. 2017), or can support rapid and organized evacuation when they are implemented along evacuation routes (Kolden and Henson 2019). Across complex landscapes, it is more effective in the long-term to prioritize fuel treatments that maximize benefits across large areas and over long time frames, rather than constrain them to the WUI.

Can wildfires, on their own, do the work of fuel treatments?

The use of managed wildfires and co-managing incidents (e.g., suppressing in some areas, and allowing other areas to burn) is increasingly promoted in the scientific literature (Stephens et al. 2016, Moreira et al. 2020). Managed wildfires are particularly appropriate in backcountry areas where lack of road access, steep topography, firefighter safety concerns, or management designations limit opportunities for active management (Hessburg et al. 2016, Huffman et al. 2020). However, in many cases the effects of fire exclusion on increased tree density, layering, surface fuels, and fuel ladders are extensive (Meyer 2015). Under these conditions, opportunities for cultural burning, prescribed burning, and managed wildfires are limited to days with low to moderate fire weather, and these windows of opportunity are shrinking under climate change (Westerling et al. 2016).

For the past several decades, land managers have generally followed one of two strategies to respond to wildfires in wNA forests. First, most agencies in the United States and Canada have followed a policy of aggressive fire suppression, and this approach is increasingly used in Mexico (Stephens and Fulé 2005). Under this policy, a small fraction of fires that escape suppression (<3%) are responsible for over 90% of area burned, based on a 1992 to 2015 reference period (Abatzoglou et al. 2018). Second, some land managers, including those managing national parks and wilderness areas, have designated large, remote areas where most wildfires are allowed to burn under moderate fire weather and fuel conditions (Huffman et al. 2020). These are termed managed wildfires, with the goal of restoring more characteristic fire regimes and landscape patterns in the context of incident-specific objectives (Table 2).

In contrast, unplanned fires that escape suppression in fire-excluded landscapes during extreme fire weather do not generally restore forest resilience. Landscapes that are consistently managed with active fire suppression

typically have a greater area burned at higher severity than those managed to restore more resilient fire regimes (Stevens et al. 2017, Rodman et al. 2020). In fire-excluded forest landscapes, forest surface and canopy fuels tend to be highly elevated, and despite active fire suppression, forests may eventually burn under extreme fire weather, which is becoming more frequent as the climate warms. For example, Povak et al. (2020) found fire severity during the 2013 Rim Fire was higher in the Stanislaus National Forest, much of which had not burned for >80 yr, compared to Yosemite National Park where past burn mosaics existed. High-severity burn patches in fires that escaped suppression are larger and less complex than in fires managed with less aggressive suppression tactics (Stevens et al. 2017), and seed sources for forest regeneration are more often distant, yielding sparse or non-existent tree regeneration (Shive et al. 2018, Korb et al. 2019, Stevens-Rumann and Morgan 2019). In dry pine and moist mixed-conifer forests, subsequent shrub establishment can lead to a cycle of repeated high-severity fires that perpetuates shrub dominance and a potentially long-term shift in alternative stable states (Collins et al. 2009, Cocking et al. 2014, Coppoletta et al. 2016, Coop et al. 2020).

Where managers allow managed wildfires to burn under prescription, burned areas are typically smaller and have greater proportions of low- and moderate-severity burn patches within the fire perimeter, and high-severity patches are typically smaller (Parks et al. 2014, Stevens et al. 2017). Within low- and moderate-severity burn patches, fuels are reduced, and forest structures resembling more typical historical conditions emerge (Holden et al. 2007, Huffman et al. 2018, Stoddard et al. 2020). In some forests, this includes characteristic patterns of small tree clumps and interspersed openings (Fig. 4; Kane et al. 2014, 2019, Jeronimo et al. 2019). In fire-excluded forests, a first entry with managed wildfire may not meet fuels reduction and management objectives unless allowed to burn at a severity that modifies stand structure (Huffman et al. 2017). Fire resilient landscapes are generally created by burning and reburning, in which prior fires modify the spread, intensity, and severity of subsequent fires (Prichard et al. 2017, Walker et al. 2018, Yocom et al. 2019, Koontz et al. 2020).

Promising strategies are emerging to delineate landscapes into operational units where decisions about applying managed fire can be considered before ignitions even occur (Thompson et al. 2016, Dunn et al. 2017). Managed wildfires are an important management tool and they are increasingly recognized as a vital component of adaptive management. However, relying solely on managed wildfires to achieve management objectives is not possible due to a number of factors that include current restrictions on the use of managed wildfire in the WUI or near other infrastructure, limited burn windows with moderate fire weather, and the potential negative consequences of allowing fire spread into nearby fire-excluded areas with elevated fuel loads.

Is the primary objective of fuel reduction treatments to assist in future firefighting response and containment?

In a review of fuel treatment options for interior western United States forests, Reinhardt et al. (2008) recommend that the central objective of fuel reduction treatments should not be to halt fire spread or reduce ignitions. Rather, fuel reduction treatments could be implemented to modify fire behavior and mitigate fire effects (Safford et al., 2012*a*, *b*), thereby reinforcing the initial resilience of the treated stand by further reducing fuels, introducing greater heterogeneity, and allowing firefighters to fight fires, as needed, using direct techniques (Stevens et al. 2014, Kalies and Yocom Kent 2016). Under adaptive management, fuel treatments are not designed to prevent or stop fires but to moderate fire behavior when fire inevitably returns (Calkin et al. 2014). However, there is a frequent misconception that fuel treatments should facilitate suppression and limit the size of wildfires (Table 1; Cochrane et al. 2012, Schoennagel et al. 2017).

The reasoning behind treating fuels to facilitate fire suppression activities is circular. If fuel treatments make suppression more successful, then wildland fuels continue to accumulate, creating even more hazardous conditions for the entire landscape. Inevitably, this makes subsequent suppression more difficult, and more areas will be burned in fewer, unmanageable events with greater ecological consequences (Collins et al. 2010, Calkin et al. 2015). This phenomenon has been described as "the wildland fire paradox" (Arno and Brown 1991). Rather than creating conditions where wildfire is easier to suppress, fuel treatments designed within a restoration or climate adaptation strategy are engineered to allow subsequent wildfires to burn without the need of full suppression tactics and to increase opportunities for prescribed or cultural burning.

Typical fuel reduction activities near communities illustrate the long-term consequences of using treatments with the expressed objective of suppressing future wildfires. Near communities, fuel reduction treatments are often explicitly implemented to create conditions that enhance fire suppression efficacy in both the surrounding wildland and WUI (Moghaddas and Craggs 2007). Treatment locations are selected based on criteria that involve community protection (Fleeger 2008), suppression concerns (Finney 2001), and fuel hazards (Schmidt et al. 2008), at stand and landscape scales (Chung et al. 2013). Suppression strategies are designed to use treated areas for burnout operations, anchor points for fire lines, and safe zones for firefighters. Some of the challenges associated with this approach are that burnout operations often burn at high severity (Backer et al. 2004), and most fire line and safe zone construction involves the cutting of live and dead trees and mineral soil exposure, all of which result in conditions that can facilitate the spread of invasive species where they are present or nearby, degrade archaeological-heritage sites, and actually

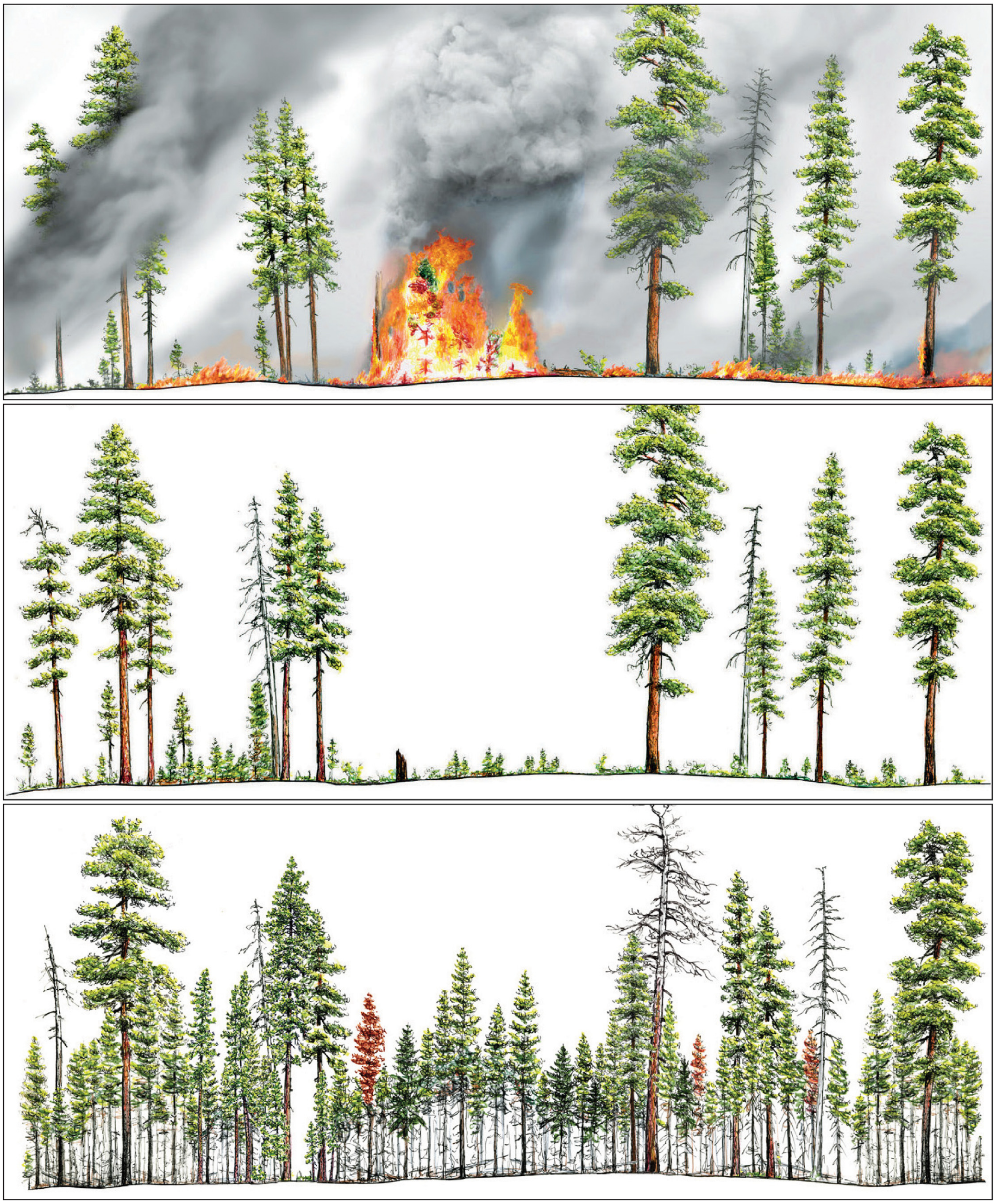


FIG. 4. Conceptual diagram of low and moderate severity fire effects on post-fire residual structure. Top: frequent fire reduces surface and ladder fuels. Middle: gradual accumulation of live and dead fuels between fires. Bottom: conditions after prolonged fire exclusion. Forest is denser and more layered, and high-severity fire is likely. Drawing credit: Robert Van Pelt.

reduce ecological resilience (Davies et al. 2010). Further, if insufficient area is treated on a landscape, the unexpected behavior of large wildfires will overwhelm the ability of small fuel treatments to facilitate effective suppression (Agee et al. 2000, Finney et al. 2001). If fuel treatments are designed such that the next wildfire can be allowed to burn with limited or no suppression, then three economic and ecological objectives might be achieved: reduced suppression costs and actions; management of future wildfires as effective fuel treatment maintenance; and favorable ecological outcomes in areas treated before wildfire.

There is little doubt that fuel reduction treatments can be effective at reducing fire severity and achieving culturally and ecologically beneficial effects, if designed and implemented correctly (Stephens et al. 2009, Fulé et al. 2012). However, fuel treatments intended only for crown fire hazard mitigation rarely constitute effective

restoration (Stephens et al. 2020). As the pace and scale of fuel treatments increase, emphasis on resilient forest structure and composition, long-term reduction of surface and canopy fuels, and adaptation to climate change are critical components of treatment objectives rather than creating conditions that are more conducive to fire suppression (Hessburg et al. 2019).

Do fuel treatments work under extreme fire weather?

Although extreme fire behavior including strong winds and column-driven fire spread can overwhelm individual treatments, there is strong scientific evidence that even under extreme weather conditions, fuel reduction treatments are effective at moderating fire severity across a range of forest types and wildfire events. For example, Walker et al. (2018) studied the 2011 Las Conchas fire in New Mexico that burned under extreme weather and found that sites that were previously prescribed burned exhibited higher conifer survival (i.e., lower severity fire) compared to sites that were not treated prior to the wildfire. Similarly, Yocom Kent et al. (2015) found that moderate- and high-severity effects in the Rodeo-Chediski Fire, which burned under extreme fire weather, were reduced from 76% in untreated areas to 57% in prescribed fire, and 38% in thin and burn treatments. Likewise, Povak et al. (2020) presented evidence that some treated areas experienced lower severity fire even under the most extreme fire growth period of the 2013 Rim Fire. Past wildfires also acted as short-term barriers to fire spread and mitigated fire severity in mixed-conifer forests of the interior western United States (Parks et al. 2015*a*, Stevens-Rumann et al. 2016). Lastly, two studies in seasonally dry mixed-conifer forests of north-central Washington State found that thinning followed by prescribed burning was an effective treatment for mitigating wildfire effects under extreme weather conditions (Prichard and Kennedy 2014, Prichard et al. 2020). Results of these observational studies are also supported by numerous modelling studies indicating that fuel treatments reduce fire intensity and effects in dry conifer forests under dry fuels and high wind speeds (Stephens and Moghaddas 2005, Ager et al. 2007, Vaillant et al. 2009, Johnson et al. 2011).

In forests characterized by moderate- and high-severity fire regimes, a limited number of studies suggest that fuel reduction treatments are ineffective at reducing fire behavior and effects, particularly under extreme weather conditions (e.g., Graham 2003, Martinson et al. 2003, Schoennagel et al. 2004). The rationale is that fires burning within moist and cold forest patches are generally controlled by climate (i.e., a warmer and drier than average year) and not controlled by fuel within patches (Turner and Romme 1994, Bessie and Johnson 1995). However, at larger spatial scales, there is strong evidence that patchwork burn mosaics resulting from reburns reduce landscape contagion, and consequently, spread and severity of wildfires, even under extreme fire weather (Stine et al. 2014, Parks et al. 2015*b*, Hessburg et al. 2016, Spies et al. 2018).

Dependent on the forest type and environmental setting, some fuel treatments are more effective at reducing adverse fire effects than others, and this can also contribute to confusion as to whether or not treatments are effective under extreme fire weather. Several studies highlight that the most effective fuel treatments include coupled thinning and burning (Kalies and Yocom Kent 2016), and emphasize the importance of retaining large, fire-resistant trees in dry mixed conifer forests (DellaSala et al. 2004, Agee and Skinner 2005, Stephens et al. 2009). Furthermore, other studies showed that fire severity decreased as wildfires progress further into areas with more treated area (Arkle et al. 2012, Kennedy and Johnson 2014), strongly suggesting that small fuel treatments or those with large perimeter-to-edge ratios are less effective than larger treatments under extreme fire weather conditions (Kennedy et al. 2019).

Finally, fuel treatments generally are designed to mitigate wildfire intensity and effects but they are not necessarily intended to impede fire spread or reduce fire size (Reinhardt et al. 2008). Consequently, when fires burn large areas under extreme fire weather some may conclude that burned-over fuel treatments were ineffective (e.g., Schoennagel et al. 2017). However, the occurrence of large fires does not necessarily suggest that existing fuel treatments were unsuccessful. Large fires have always been a part of fire-prone forests, and within large fire events fuel treatments can allow fires to continue burning but mitigate fire severity and enhance the heterogeneity of fire effects.

Is the scale of the problem too great? Can we ever catch up?

Recent meta-analyses of fuel treatment effectiveness demonstrate that at landscape and regional scales, fuel treatments account for only a small fraction (~1%) of the area burned by wildfires (e.g., Barnett et al. 2016*a*, Kolden 2019). Therefore, there is some concern that treatments are ineffective because under current prescription levels, wildfires may not actually encounter treated areas during the duration of their potential effectiveness (Odion and Hanson 2006, Rhodes and Baker 2008). While this is factually accurate at the current pace and scale of treatment in wNA, the question is not whether every wildfire can be impacted by fuels treatments, but whether treatments can be strategically used to multiply their benefits and promote greater opportunities for applying wildland fire across landscapes. The scientific evidence that fuel reduction treatments can mitigate fire behavior and effects strongly supports a conclusion that expanding treated areas, including the use of forest thinning, prescribed burning, cultural burning, and managed wildfires, will lead to greater landscape resilience to future wildfires.

Ongoing warming and drying are linked to increasing large fire occurrence, contributing to large increases in area burned (Abatzoglou and Williams 2016) and area burned as high severity (Parks and Abatzoglou 2020) in wNA in recent decades. Given projected increases in warming due to climate change, burn probability is increasing in many wNA forests (Littell et al. 2018, Hurteau et al. 2019) along with increasing likelihood that future wildfires will impact a larger proportion of landscapes. In this light, the current pace and scale of fuels treatments is insufficient to address the scale of fire exclusion. Furthermore, treated areas require ongoing maintenance to retain efficacy (Krofcheck et al. 2017, Vaillant and Reinhardt 2017), making it difficult to expand treated areas across a landscape without significant additional financial and personnel investments (North et al. 2015*a*). Thus, the scope, scale, and urgency of adapting wNA forests to climate change and future wildfires is immense.

Given the complexity of forest ecosystems, the economic and personnel investment required, and the policy and management constraints, there is no single management tool that is adequate to increase the resilience of wNA landscapes to future wildfires. Coupled thinning and burning treatments will be especially helpful in dry pine, oak woodlands, and dry mixed conifer forests, while restoration of more characteristic forest successional and nonforest patchworks using managed moderate and high severity wildfires will be key in cold forests. Forest managers in western Australia have reduced the frequency of large and severe wildfires, but only after building extensive landscape networks of strategic treatments (i.e., spatially linked naturally occurring and treated areas of reduced fuels prior to the outbreak of wildfires) and by conducting frequent prescribed burning under moderate fire weather and including Indigenous fire use over large areas (Boer et al. 2009, Sneeuwjagt et al. 2013). Similar approaches are being used in U.S. national forest, wilderness, and park areas to allow for more area of managed wildfires (Table 2). Given limitations on where mechanical thinning, prescribed and cultural burning, and managed wildfire are practical or allowed, combining these tools over broad areas can markedly expand treatment extent and reduce impact of large wildfires.

Fire hazard, burn probability, and fire ecology vary widely across wNA forest landscapes. Prior knowledge of cultural burning practices, ignition and weather patterns, vegetation and fuel distributions, and topography all provide critical information for prioritizing fuel treatments in areas with the highest risk of burning (Ager et al. 2010, 2016). Near population centers, humans are often responsible for the majority of wildfire ignitions, and they provide ignition sources in highly predictable areas and seasons of the year, when natural ignitions are rare (Balch et al. 2017, Keeley and Syphard 2018). Ignition pattern and frequency interact with fuels, weather, and topography to influence fire occurrence, leading to heterogeneous burn probabilities across a landscape (Ager et al. 2012, Povak et al. 2018). Using prior knowledge of human and lighting-caused fire starts coupled with knowledge of the probability of fire spread and likely severity, managers can identify the areas of any landscape where uncharacteristic or impactful fires will likely occur (Parisien and Moritz 2009, Parisien et al. 2012), and decrease the proportion of the landscape that requires treatment.

There are a number of available tools and approaches to identify areas that would benefit from strategically placed fuel treatments. In general, fuel treatments are not implemented at random, and for good reason (Finney et al. 2007). A comparison of random vs. strategically placed treatments showed that a significant reduction in area could be achieved with strategic placement (Ager et al. 2013, 2016), where that opportunity exists. Quantifying the probability of high-severity wildfire across a given landscape and focusing thinning treatments on high-probability areas can decrease the required treatment area by >50% (Krofcheck et al. 2019). However, the success of these strategies depends on maintaining the treatments and reintroducing fire to a larger portion of the landscape (Agee and Skinner 2005, Barros et al. 2018). Where reserved areas are abundant or widely distributed, opportunities for spatially optimizing fuel treatments are limited, and considerably more treated area may be required outside of reserves (Finney et al. 2007).

In summary, justifying inaction based on the scale of the problem is too large is highly circular. Evidence supports increasing the pace of treatments to significantly reduce the area impacted by uncharacteristic wildfire, even under a changing climate (Liang et al. 2018). For example, managers can expand areas where burn prescriptions are applied to reduce fuels and increase forest heterogeneity (Safford et al. 2012*a*,*b*, Striplin et al. 2020). The efficacy of these was historically demonstrated by Indigenous burning practices that amplified natural lightning ignitions in many seasonally dry forests, thereby modifying active fire regimes and fire effects, and diversifying the seasonality and frequency of fires (Crawford et al. 2015, Trauernicht et al. 2015, Taylor et al. 2016). Managed wildfires can also increase forest and fuel heterogeneity, constraining subsequent fire size and severity (Collins et al. 2009, Parks et al. 2015*b*, Barros et al. 2018). When used in conjunction with mechanical treatments and prescribed or cultural burning, managed wildfire presents an opportunity to increase the effectiveness of treatments across large landscapes (North et al. 2012).

Will planting more trees in wNA forests help to mitigate climate change?

Tree plantations have long been a debated aspect of forest management, and more recently, climate change mitigation (Alig 1997, Chmura et al. 2011). Planting

after harvest to increase forest productivity were the central justifications for past clearcut logging, even as a growing body of science demonstrated that plantations (1) did not provide the needed ecological structures or functional diversity of old-growth forests, (2) were not necessarily more productive than mature forests (Franklin et al. 2002), and (3) without surface fuel treatment, could be conducive to high-severity wildfires (Thompson et al. 2007). Similarly, planting seedlings after post-fire salvage logging is sometimes used to expedite tree regeneration following high-severity fire. Without strategic management, post-fire plantations may be overstocked, dominated by a single species (North et al. 2019), lack tree clumping and canopy gaps, and pose significant wildfire hazard (Kobziar et al. 2009), particularly without post-harvest slash reduction (Donato et al. 2009).

A recent proposal to combat climate change includes planting a trillion trees globally, including substantial reforestation in the western United States (Bastin et al. 2019). The study suggested that these additional trees would sequester sufficient atmospheric carbon to curb climate change. Baseline assumptions and findings from this study have been contested by scientists (Veldman et al. 2019, Holl and Brancalion 2020) as the study failed to account for forest interactions with climate, drought, and wildfire dynamics. In addition to future disturbance resilience, numerous other barriers currently impede large-scale reforestation efforts (Fargione et al. 2021).

Across wNA, most of the forest carbon is captured in moist temperate forests with high precipitation levels and net primary productivity, including the coastal ranges along the Pacific Coast, western Cascade and western Sierra Nevada Mountain Ranges (Hudiburg et al. 2009). These forests possess complex, heterogeneous structures, some of which developed with infrequent wildfires. Others, including those in southwestern Oregon and northern California, were also influenced by a long legacy of Indigenous burning (Anderson 2013, Merschel et al. 2014). Because most of the standing biomass in high productivity wNA forests occurs in live trees, when these forests burn, relatively low levels of carbon are initially emitted, with most of the biomass retained either in standing trees and snags or to newly downed heavy fuels that slowly release carbon to the atmosphere through decomposition, unless they subsequently burn in a reburn fire event (Stenzel et al. 2019, Lutz et al. 2020). By contrast, even-aged stands, both naturally occurring (e.g., lodgepole pine forests) and in young plantations, are relatively homogeneous in structure, and with elevated surface fuels, can facilitate high-intensity, severe fire (Bowman et al. 2019). Climate change-induced shortening of fire return intervals may ultimately convert some of these live carbon pools from sinks to sources (Turner et al. 2019, Foster et al. 2020).

In fire-adapted dry mixed conifer forests, dense tree plantations are highly susceptible to future wildfires and drought. However, a promising approach to retaining and sequestering carbon in dry, fire-prone forests is to retain existing large-diameter trees and restore characteristic low-severity fire to maintain low-severity fire to maintain resilient forest structure and composition (Hurteau and North 2009). It is still debatable whether prescribed burning and removal of small diameter trees and ladder fuels will actually increase or decrease aboveground carbon stores (Campbell et al. 2012, Restaino and Peterson 2013) and is likely site dependent, but there is broad scientific agreement that these management actions are key to increasing forest ecological resilience, which ultimately stabilizes forest carbon stocks (Hurteau et al. 2019, Krofcheck et al. 2019, Westlind and Kerns 2021). Managed landscape mosaics will be particularly critical to maintaining legacy old-growth forests and minimizing sink-to-source conversions due to fire and other disturbances (Barbero et al. 2015, Liang et al. 2017). Finally, governmental cap-and-trade and carbon taxation programs must accurately account for the complex role fire plays in carbon cycle feedbacks and carbon maintenance, rather than simply characterizing fire as a net carbon loss (Hurteau et al. 2008, North et al. 2009).

Across wNA forests, tree planting can serve as an important tool to nudge the trajectory of post-fire landscapes towards more climate-adapted tree species or genotypes, particularly in areas where seed source is limited (North et al. 2019). However, traditional high density plantations will often predispose forests to high-severity fire where pre-commercial thinning and associated fuel treatments are not implemented, which is increasingly the case (McCarley et al. 2017). Alternatives to traditional plantations are emerging that are designed to promote resilience to future fire and drought from the beginning of the planting process. These include planting drought-conditioned seedlings reared from lower-elevation seed stock, planting discontinuous "founder stands" or "nucleation islands" of trees into portions of stand-replacing patches far from tree refugia, and planning for the reintroduction of fire into younger planted stands as they develop (Peterson et al. 2007, Landis et al. 2011).

Is post-fire management needed or even ecologically justified?

Many contemporary wildfires exhibit a range of post-fire effects (Thode et al. 2011); variable sized patches of stand-replacing or partial stand replacing fire are embedded within a matrix of live forest (Stevens et al. 2017). Among large fires, these patches of stand-replacing fire may themselves contain isolated and variably sized patches of live trees often referred to as fire refugia (Meddens et al. 2018, Krawchuk et al. 2020). Thus, the post-fire landscape can be viewed as a complex patchwork of interconnected surviving forest, the product of low and moderate severity fires, high-severity patches, and isolated refugia (Coop et al. 2019). However, these post-fire landscapes are not necessarily on

resilient trajectories. Fire refugia may be in uncharacteristic locations, and active forest and fuels management are often required after the fire to promote future forest resilience to disturbance and climate change and to protect valued cultural resources.

Patches of low- and moderate-severity fire generally have short-term resistance to future fire due to the reduction of surface fuels from the first burn (Prichard et al. 2017). Compared to low-severity fire, moderate-severity fire events can create a residual stand structure that more closely approximates historical conditions (Collins et al. 2011, Huffman et al. 2017). However, moderate-severity fires that burn through previously dense forest also leave considerable standing and down wood, which can lead to elevated fuel loads and high-severity fire in subsequent reburns (Collins et al. 2018). Thus, post-fire fuel reduction of the trees that encroached during the period of fire exclusion can be warranted to improve the fire resilience of residual forests, including fire refugia.

Smaller refugial patches within larger burned patches are increasingly recognized as having significant cultural and ecological value by preserving biological and cultural legacies that can contribute to forest succession via seed dispersal (Johnstone et al. 2016, Meddens et al. 2018). Small refugia in particular make disproportionate contributions to reforestation potential within larger patches of stand-replacing fire (Shive et al. 2018, Coop et al. 2019). However, isolated tree refugia can have a significant standing and down fuel component around their edges due to adjacent high-severity burn effects (Lydersen et al. 2019*a*). Given their outsized importance as biological legacies, surface fuel reduction to "harden" the edges of refugia may be critical to their future resilience and prioritize refugia retention during wildland firefighting operations (Meddens et al. 2018).

Large patches of stand-replacing fire are an increasing focus of research (Coop et al. 2020). Independent of subsequent fire dynamics, regeneration is challenged by seed dispersal limitations and a warming climate (Stevens-Rumann and Morgan 2019). Fuel conditions in large patches of stand-replacing fire are usually dominated by coarse wood, regenerating shrubs, and hardwoods, increasing the risk of subsequent high-severity, and occurrence of long-duration re-burns (Coppoletta et al. 2016, Prichard et al. 2017). Collectively, these conditions pose a substantial management challenge if the objective is to restore at least a portion of large burn patches to conifer forest, as this is unlikely over decades to centuries without management intervention (Coop et al. 2020).

Fuels management and regeneration dynamics in stand-replacing patches are closely related. In high-severity patches, management to reduce coarse wood accumulations and flammable shrubs may promote post-fire tree regeneration and mitigate future fire severity (Peterson et al. 2015, Lydersen et al. 2019*b*). In planted forests, coarse wood presents a different challenge, as downed logs facilitate seedling survival through shading and moisture retention (Castro et al. 2011) but pose a risk to seedlings if they burn (Peterson et al. 2015). Understanding the range and variability of historical reburning would provide essential guidance of restoration targets to improve the post-fire resilience of regenerating landscapes.

Strategic tree planting can be used to encourage the re-establishment of some post-fire landscapes and for climate change adaptation, particularly where conditions are not favorable to natural regeneration (see previous question). Post-fire mechanical thinning (e.g., salvage logging) is often driven by economic and safety considerations but may have some ecological benefits in terms of reduced future surface fuel loads and fire hazard 10–20 yr post-fire (Peterson et al. 2015). Future research in this area is warranted to investigate the impacts of variable density harvests and how potential ecological tradeoffs vary over time (e.g., Ritchie et al. 2013).

Conclusions

During this time of rapid environmental change, the impacts of climatic changes on forests and their associated fire regimes cannot be overstated. In addition to the increased incidence of large wildfires, tree mortality associated with persistent drought and die-off events, chronic forest insect outbreaks, and increasingly common tree regeneration failures are all critical management considerations (Stephens et al. 2016, Coop et al. 2020). In a majority of cases, forest management and fuel reduction treatments will not return landscapes to any historical condition or fire regime, nor is that a particularly useful premise on which to base adaptive forest management (Allen et al. 2011, Hanberry et al. 2015, Falk et al. 2019). Instead, intentional management focused on adapting current forest conditions to a rapidly evolving future climate is needed. Adaptations can foster forest resilience to longer, warmer, drier, and windier fire seasons, increasing incidence of episodic, multi-year to decadal droughts, and increasing dominance of severe wildfire and insect disturbances. Given the rapid increase in human-caused large wildfires, mitigating unplanned human ignitions is another critical wildland fire management issue (Balch et al. 2017), that by itself can reshape wildfire and forest landscape futures.

Although the management situation for wNA forests is daunting, our review of the scientific literature offers clear guidance. In seasonally dry wNA forests that were historically dominated by fire-resistant species, restoring open, fire-tolerant canopy structure and composition, favoring larger tree sizes, and reducing surface fuels can effectively mitigate subsequent wildfire and stabilize carbon stocks (Fig. 1). In many instances, these adaptation actions, with ongoing maintenance, will also enable future wildfire events to continually reinforce resilient structure, composition, and fuels.

Ecological departures associated with fire exclusion are not confined to seasonally dry pine and mixed-conifer forests. Across a wide range of wNA forests, landscape-level treatment prescriptions that promote resilient patchworks with heterogeneous nonforest and forest ages can reduce the extent of high-severity wildfires and make landscapes less susceptible to extensive insect and disease outbreaks. Restoration of fire resilient mosaics in moist mixed-conifer forests, mixed conifer-hardwood forests, fire-prone deciduous forests (e.g., aspen), and cold forests is also needed.

Despite calls to restore fire as a cultural and ecological process (e.g., The U.S. National Wildland Fire Cohesive Strategy), the dominant approach to wildfire management continues to be aggressive suppression. Response to unplanned fire starts is highly successful in the United States and Canada and is becoming increasingly common in Mexico. However, a small fraction of fires that escape suppression (2–3%) generally burn under extreme fire weather conditions, lead to explosive fire growth, and account for >90% of annual area burned (Abatzoglou et al. 2018). The strategy to actively suppress fire is a highly consequential active management prescription, with surface and canopy *fuel accumulation* as a consequence. Continued forest infilling and fuel accumulation predisposes forests to high-severity fire when fire inevitably returns (North et al. 2015*b*).

Not surprisingly, recommendations to increase wNA forest resilience to climate change and wildfires are in close alignment with Indigenous knowledge, cultural resource values, and desired land management strategies (Kimmerer and Lake 2001, Lake et al. 2018, Roos et al. 2021). Over millennia, Indigenous burning practices influenced fire regimes, which contributed to the resilient composition and structure of many historical wNA forest and nonforest ecosystems. Although European colonization severely curtailed and displaced Indigenous land management (Lake et al. 2017, Lake and Christianson 2019), Indigenous knowledge for the maintenance of fire-dependent ecosystems and services endures (Huffman 2013). Given the urgent need for adaptive forest management in the 21st-century, an intentional merging of Indigenous and western knowledge is needed to guide future forest conditions and restore active fire regimes to wNA forests.

Acknowledgments

This synthesis project was funded by the US Forest Service Pacific Northwest and Pacific Southwest Research Stations (P. F. Hessburg, N. A. Povak), California Department of Forestry and Fire Protection (S. J. Prichard, R. K. Hagmann, P. Khatri-Chhetri); Ecological Restoration Institute (D. Huffman, R. K. Hagmann); Washington State Department of Natural Resources (D. Churchill, R. K. Hagmann); the Wilderness Society (R. K. Hagmann); Nature Conservancy–Oregon (R. K. Hagmann); and Conservation Northwest (R. K. Hagmann). We thank Mike Battaglia, Ellis Margolis, and James Rosen for their constructive reviews and the U.S. Fish and Wildlife Service for assistance with publication. The authors also wish to acknowledge NSF's Growing Convergence Research Program (Award Number 2019762) for support of this work.

Literature Cited

Abatzoglou, J. T., and A. P. Williams. 2016. Impact of anthropogenic climate change on wildfire across western US forests. Proceedings of the National Academy of Sciences USA 113:11770–11775.

Abatzoglou, T. T., J. K. Balch, B. A. Bradley, and C. A. Kolden. 2018. Human-related ignitions concurrent with high winds promote large wildfires across the USA. International Journal of Wildland Fire 27:377–386.

Agee, J. K. 1996. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C., USA.

Agee, J. K., B. Bahro, M. A. Finney, P. N. Omi, D. Sapsis, C. Skinner, J. van Wagtendonk, and C. P. Weatherspoon. 2000. The use of shaded fuelbreaks in landscape fire management. Forest Ecology and Management 127:55–66.

Agee, J., and C. N. Skinner. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211:83–96.

Ager, A. A., M. A. Day, K. C. Short, and C. R. Evers. 2016. Assessing the impacts of federal forest planning on wildfire risk mitigation in the Pacific Northwest, USA. Landscape and Urban Planning 147:1–17.

Ager, A., A. J. McMahan, J. J. Barrett, and C. W. McHugh. 2007. A simulation study of thinning and fuel treatments on a wildland–urban interface in eastern Oregon, USA. Landscape and Urban Planning 80:292–300.

Ager, A. A., N. M. Vaillant, and M. A. Finney. 2010. A comparison of landscape fuel treatment strategies to mitigate wildland fire risk in the urban interface and preserve old forest structure. Forest Ecology and Management 259:1556–1570.

Ager, A. A., N. M. Vaillant, M. A. Finney, and H. K. Preisler. 2012. Analyzing wildfire exposure and source–sink relationships on a fire prone forest landscape. Forest Ecology and Management 267:271–283.

Ager, A. A., N. M. Vaillant, and A. McMahan. 2013. Restoration of fire in managed forests: a model to prioritize landscapes and analyze tradeoffs. Ecosphere 4:19.

Alig, R. 1997. Assessing effects of mitigation strategies for global climate change with an intertemporal model of the U.S. forest and agriculture sectors. Environmental and Resource Economics 9:259–274.

Allen, C. D., et al. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management 259:660–684.

Allen, C. R., J. J. Fontaine, K. L. Pope, and A. S. Garmestani. 2011. Adaptive management for a turbulent future. Journal of Environmental Management 92:1339–1345.

Anderson, M. K. 2013. Trending the Wild: Native American knowledge and the management of California's natural resources. University of California Press, Berkeley, California, USA.

Arkle, R. S., D. S. Pilliod, and J. L. Welty. 2012. Pattern and process of prescribed fires influence effectiveness at reducing wildfire severity in dry coniferous forests. Forest Ecology and Management 276:174–184.

Arno, S. F., and J. K. Brown. 1991. Overcoming the paradox in managing wildland fire. National Emergency Training Center, Emmitsburg, Maryland, USA.

Backer, D. M., S. E. Jensen, and G. R. McPherson. 2004. Impacts of fire-suppression activities on natural communities. Conservation Biology 18:937–946.

Baker, W. L., and C. T. Hanson. 2017. Improving the use of early timber inventories in reconstructing historical dry

forests and fire in the western United States. Ecosphere 8: e01935.

Balch, J. K., B. A. Bradley, J. T. Abatzoglou, R. C. Nagy, E. J. Fusco, and A. L. Mahood. 2017. Human-started wildfires expand the fire niche across the United States. Proceedings of the National Academy of Sciences USA 114:2946–2951.

Bar Massada, A., V. C. Radeloff, and S. I. Stewart. 2011. Allocating fuel breaks to optimally protect structures in the wildland–urban interface. International Journal of Wildland Fire 20:59.

Barbero, R., J. T. Abatzoglou, N. K. Larkin, C. A. Kolden, and B. Stocks. 2015. Climate change presents increased potential for very large fires in the contiguous United States. International Journal of Wildland Fire 24:892–899.

Barnett, K., C. Miller, and T. J. Venn. 2016*a*. Using risk analysis to reveal opportunities for the management of unplanned ignitions in wilderness. Journal of Forestry 114:610–618.

Barnett, K., S. A. Parks, C. Miller, and H. T. Naughton. 2016*b*. Beyond fuel treatment effectiveness: characterizing interactions between fire and treatments in the US. Forests 7:237.

Barnhart, S. J., J. R. McBride, and P. Warner. 1996. Invasion of northern oak woodlands by *Pseudotsuga menziesii* (Mirb.) Franco in the Sonoma Mountains of California. Madroño 43:28–45.

Barros, A. M., A. A. Ager, M. A. Day, M. A. Krawchuk, and T. A. Spies. 2018. Wildfires managed for restoration enhance ecological resilience. Ecosphere 9:e02161.

Bastin, J.-F., Y. Finegold, C. Garcia, D. Mollicone, M. Rezende, D. Routh, C. M. Zohner, and T. W. Crowther. 2019. The global tree restoration potential. Science 365:76–79.

Battaglia, M. A., B. Gannon, P. M. Brown, P. J. Fornwalt, A. S. Cheng, and L. S. Huckaby. 2018. Changes in forest structure since 1860 in ponderosa pine dominated forests in the Colorado and Wyoming Front Range, USA. Forest Ecology and Management 422:147–160.

Battaglia, M. A., and W. D. Shepperd. 2007. Ponderosa pine, mixed conifer, and spruce-fir forests [Chapter 2]. Fire ecology and management of the major ecosystems in southern Utah. RMRS-GTR-202. U.S. Forest Service, Fort Collins, Colorado, USA.

Belote, R. T., A. J. Larson, and M. S. Dietz. 2015. Tree survival scales to community-level effects following mixed-severity fire in a mixed conifer forest. Forest Ecology and Management 353:221–231.

Bentz, B. J., J. Régnière, C. J. Fettig, E. M. Hansen, J. L. Hayes, J. A. Hicke, R. G. Kelsey, J. F. Negrón, and S. J. Seybold. 2010. Climate change and bark beetles of the western United States and Canada: direct and indirect effects. BioScience 60:602–613.

Bessie, W. C., and E. A. Johnson. 1995. The relative importance of fuels and weather on fire behavior in subalpine forests. Ecology 76:747–762.

Bigelow, S. W., and M. P. North. 2012. Microclimate effects of fuels-reduction and group-selection silviculture: Implications for fire behavior in Sierran mixed-conifer forests. Forest Ecology and Management 264:51–59.

Bladon, K. D. 2018. Rethinking wildfires and forest watersheds. Science 359:1001–1002.

Boer, M. M., R. J. Sadler, R. S. Wittkuhn, L. McCaw, and P. F. Grierson. 2009. Long-term impacts of prescribed burning on regional extent and incidence of wildfires—evidence from 50 years of active fire management in SW Australian forests. Forest Ecology and Management 259:132–142.

Boisramé, G., S. Thompson, B. Collins, and S. Stephens. 2017. Managed wildfire effects on forest resilience and water in the Sierra Nevada. Ecosystems 20:717–732.

Bowman, D. M., et al. 2019. Human–environmental drivers and impacts of the globally extreme 2017 Chilean fires. Ambio 48:350–362.

Bowman, D. M. J. S., C. A. Kolden, J. T. Abatzoglou, F. H. Johnston, G. R. van der Werf, and M. Flannigan. 2020. Vegetation fires in the Anthropocene. Nature Reviews Earth & Environment 1:505–515.

Bradley, C. M., C. T. Hanson, and D. A. DellaSala. 2016. Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States? Ecosphere 7:e01492.

Brown, P. M., C. Gentry, and Q. Yao. 2019. Historical and current fire regimes in ponderosa pine forests at Zion National Park, Utah: restoration of pattern and process after a century of fire exclusion. Forest Ecology and Management 445:1–12.

Brown, R. T., J. K. Agee, and J. F. Franklin. 2004. Forest restoration and fire: principles in the context of place. Conservation Biology 18:903–912.

Calkin, D. E., J. D. Cohen, M. A. Finney, and M. P. Thompson. 2014. How risk management can prevent future wildfire disasters in the wildland-urban interface. Proceedings of the National Academy of Sciences USA 111:746–751.

Calkin, D. E., M. P. Thompson, and M. A. Finney. 2015. Negative consequences of positive feedbacks in US wildfire management. Forest Ecosystems 2:9.

Campbell, J. L., M. E. Harmon, and S. R. Mitchell. 2012. Can fuel-reduction treatments really increase forest carbon storage in the western US by reducing future fire emissions? Frontiers in Ecology and the Environment 10:83–90.

Castro, J., C. D. Allen, M. Molina-Morales, S. Maranon-Jimenez, A. Sanchez-Miranda, and R. Zamora. 2011. Salvage logging versus the use of burnt wood as a nurse object to promote post-fire tree seedling establishment. Restoration Ecology 19:537–544.

Chmura, D. J., P. D. Anderson, G. T. Howe, C. A. Harrington, J. E. Halofsky, D. L. Peterson, D. C. Shaw, and J. Brad St.Clair. 2011. Forest responses to climate change in the northwestern United States: ecophysiological foundations for adaptive management. Forest Ecology and Management 261:1121–1142.

Chung, W., G. Jones, K. Krueger, J. Bramel, and M. Contreras. 2013. Optimising fuel treatments over time and space. International Journal of Wildland Fire 22:1118.

Churchill, D. J., G. C. Carnwath, A. J. Larson, and S. A. Jeronimo. 2017. Historical forest structure, composition, and spatial pattern in dry conifer forests of the western Blue Mountains, Oregon. General Technical Report PNW-GTR-956. USDA Forest Service, Portland, Oregon, USA.

Churchill, D. J., A. J. Larson, M. C. Dahlgreen, J. F. Franklin, P. F. Hessburg, and J. A. Lutz. 2013. Restoring forest resilience: from reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management 291:442–457.

Cochrane, M. A., C. J. Moran, M. C. Wimberly, A. D. Baer, M. A. Finney, K. L. Beckendorf, J. Eidenshink, and Z. Zhu. 2012. Estimation of wildfire size and risk changes due to fuels treatments. International Journal of Wildland Fire 21:357–367.

Cocking, M. I., J. M. Varner, and E. E. Knapp. 2014. Long-term effects of fire severity on oak-conifer dynamics in the southern Cascades. Ecological Applications 24:94–107.

Collins, B. M., R. G. Everett, and S. L. Stephens. 2011. Impacts of fire exclusion and recent managed fire on forest structure in old growth Sierra Nevada mixed-conifer forests. Ecosphere 2:1–14.

Collins, B. M., J. M. Lydersen, R. G. Everett, and S. L. Stephens. 2018. How does forest recovery following

moderate-severity fire influence effects of subsequent wildfire in mixed-conifer forests? Fire Ecology 14:3.

Collins, B. M., J. D. Miller, A. E. Thode, M. Kelly, J. W. van Wagtendonk, and S. L. Stephens. 2009. Interactions among wildland fires in a long-established Sierra Nevada natural fire area. Ecosystems 12:114–128.

Collins, B. M., S. L. Stephens, J. J. Moghaddas, and J. Battles. 2010. Challenges and approaches in planning fuel treatments across fire-excluded forested landscapes. Journal of Forestry 108:24–31.

Congress.gov. 2020. H.R.5859 - 116th congress (2019-2020): Trillion Trees Act. https://www.congress.gov/bill/116th-congress/house-bill/5859

Coop, J. D., et al. 2020. Wildfire-driven forest conversion in western North American landscapes. BioScience 70:659–673.

Coop, J. D., T. J. DeLory, W. M. Downing, S. L. Haire, M. A. Krawchuk, C. Miller, M.-A. Parisien, and R. B. Walker. 2019. Contributions of fire refugia to resilient ponderosa pine and dry mixed-conifer forest landscapes. Ecosphere 10:e02809.

Coppoletta, M., K. E. Merriam, and B. M. Collins. 2016. Post-fire vegetation and fuel development influences fire severity patterns in reburns. Ecological Applications 26:686–699.

Crawford, J. N., S. A. Mensing, F. K. Lake, and S. R. H. Zimmerman. 2015. Late Holocene fire and vegetation reconstruction from the western Klamath Mountains, California, USA: a multi-disciplinary approach for examining potential human land-use impacts. Holocene 25:1341–1357.

Crockett, J. L., and A. L. Westerling. 2017. Greater temperature and precipitation extremes intensify western U.S. droughts, wildfire severity, and Sierra Nevada tree mortality. Journal of Climate 31:341–354.

Daniels, S. E., and G. B. Walker. 1995. Managing local environmental conflict amidst national controversy. International Journal of Conflict Management 6:290–311.

Davies, G. M., A. A. Smith, A. J. MacDonald, J. D. Bakker, and C. J. Legg. 2010. Fire intensity, fire severity and ecosystem response in heathlands: factors affecting the regeneration of *Calluna vulgaris*. Journal of Applied Ecology 47:356–365.

DellaSala, D. A., R. L. Hutto, C. T. Hanson, M. L. Bond, T. Ingalsbee, D. Odion, and W. L. Baker. 2017. Accommodating mixed-severity fire to restore and maintain ecosystem integrity with a focus on the Sierra Nevada of California, USA. Fire Ecology 13:148–171.

DellaSala, D. A., J. E. Williams, C. D. Williams, and J. F. Franklin. 2004. Beyond smoke and mirrors: a synthesis of fire policy and science. Conservation Biology 18:976–986.

Donato, D. C., J. B. Fontaine, J. L. Campbell, W. D. Robinson, J. B. Kauffman, and B. E. Law. 2009. Conifer regeneration in stand-replacement portions of a large mixed-severity wildfire in the Klamath-Siskiyou Mountains. Canadian Journal of Forest Research 39:823–838.

Dubay, T., D. Egan, E. E. Hjerpe, W. Selig, D. Brewer, D. Coelho, Z. Wurtzebach, C. Schultz, and A. E. M. Waltz. 2013. Breaking barriers, building bridges: collaborative forest landscape restoration handbook. Northern Arizona University, Flagstaff, Arizona, USA.

Dunn, C. J., and J. D. Bailey. 2016. Tree mortality and structural change following mixed-severity fire in *Pseudotsuga* forests of Oregon's western Cascades, USA. Forest Ecology and Management 365:107–118.

Dunn, C. J., M. P. Thompson, and D. E. Calkin. 2017. A framework for developing safe and effective large-fire response in a new fire management paradigm. Forest Ecology and Management 404:184–196.

Eisenberg, C., et al. 2019. Out of the ashes: ecological resilience to extreme wildfire, prescribed burns, and indigenous burning in ecosystems. Frontiers in Ecology and Evolution 7:436.

Engber, E. A., J. M. Varner, L. A. Arguello, and N. G. Sugihara. 2011. The effects of conifer encroachment and overstory structure on fuels and fire in an oak woodland landscape. Fire Ecology 7:32–50.

Estes, B. L., E. E. Knapp, C. N. Skinner, and F. C. C. Uzoh. 2012. Seasonal variation in surface fuel moisture between unthinned and thinned mixed conifer forest, northern California, USA. International Journal of Wildland Fire 21:428–435.

Falk, D. A., A. C. Watts, and A. E. Thode. 2019. Scaling ecological resilience. Frontiers in Ecology and Evolution 7:275.

Fargione, J., et al. 2021. Challenges to the reforestation pipeline in the United States. Frontiers in Forests and Global Change 4:1–8.

Finney, M. A. 2001. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. Forest Science 47:219–228.

Finney, M. A. 2007. A computational method for optimizing fuel treatment locations. International Journal of Wildland Fire 16:702–711.

Fleeger, W. E. 2008. Collaborating for success: community wildfire protection planning in the Arizona White Mountains. Journal of Forestry 106:78–82.

Foster, D. L., J. J. Battles, B. M. Collins, R. A. York, and S. L. Stephens. 2020. Potential wildfire and carbon stability in frequent-fire forests in the Sierra Nevada: trade-offs from a long-term study. Ecosphere 11:e03198.

Franklin, J. F., et al. 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management 155:399–423.

Franklin, J. F., and K. N. Johnson. 2012. A restoration framework for federal forests in the Pacific Northwest. Journal of Forestry 110:429–439.

Freeman, J., L. Kobziar, E. W. Rose, and W. Cropper. 2017. A critique of the historical-fire-regime concept in conservation. Conservation Biology 31:976–985.

Fulé, P. Z., et al. 2014. Unsupported inferences of high-severity fire in historical dry forests of the western United States: response to Williams and Baker. Global Ecology and Biogeography 23:825–830.

Fulé, P. Z., J. E. Crouse, J. P. Roccaforte, and E. L. Kalies. 2012. Do thinning and/or burning treatments in western USA ponderosa or Jeffrey pine-dominated forests help restore natural fire behavior? Forest Ecology and Management 269:68–81.

Fulé, P. Z., and D. C. Laughlin. 2007. Wildland fire effects on forest structure over an altitudinal gradient, Grand Canyon National Park, USA. Journal of Applied Ecology 44:136–146.

Fulé, P. Z., A. E. Waltz, W. W. Covington, and T. A. Heinlein. 2001. Measuring forest restoration effectiveness in reducing hazardous fuels. Journal of Forestry 11:24–29.

Goodwin, M. J., M. P. North, H. S. J. Zald, and M. D. Hurteau. 2018. The 15-year post-treatment response of a mixed-conifer understory plant community to thinning and burning treatments. Forest Ecology and Management 429:617–624.

Goodwin, M. J., M. P. North, H. S. J. Zald, and M. D. Hurteau. 2020. Changing climate reallocates the carbon debt of frequent-fire forests. Global Change Biology 26:6180–6189.

Graham, R. T. 2003. Hayman fire case study. General Technical Report RMRS-GTR-114. USDA Forest Service, Ogden, Utah, USA.

Hagmann, R. K., et al. 2021. Evidence for widespread changes in western North American forest structure, composition, and wildfire regimes. Ecological Applications.

Hagmann, R. K., J. F. Franklin, and K. N. Johnson. 2014. Historical conditions in mixed-conifer forests on the eastern slopes of the northern Oregon Cascade Range, USA. Forest Ecology and Management 330:158–170.

Hallema, D. W., G. Sun, P. V. Caldwell, S. P. Norman, E. C. Cohen, Y. Liu, K. D. Bladon, and S. G. McNulty. 2018. Burned forests impact water supplies. Nature Communications 9:1307.

Hanan, E. J., J. Ren, C. L. Tague, C. A. Kolden, J. T. Abatzoglou, R. R. Bart, M. C. Kennedy, M. Liu, and J. Adam. 2020. How climate change and fire exclusion drive wildfire regimes at actionable scales. Environmental Research Letters 16:024051.

Hanberry, B. B., R. F. Noss, H. D. Safford, S. K. Allison, and D. C. Dey. 2015. Restoration is preparation for the future. Journal of Forestry 113:425–429.

Hanson, C. T., and D. C. Odion. 2014. Is fire severity increasing in the Sierra Nevada, California, USA? International Journal of Wildland Fire 23:1–8.

Harrod, R. J., D. W. Peterson, N. A. Povak, and E. K. Dodson. 2009. Thinning and prescribed fire effects on overstory tree and snag structure in dry coniferous forests of the interior Pacific Northwest. Forest Ecology and Management 258:712–721.

Hartsough, B. R., S. Abrams, R. J. Barbour, E. S. Drews, J. D. McIver, J. J. Moghaddas, D. W. Schwilk, and S. L. Stephens. 2008. The economics of alternative fuel reduction treatments in western United States dry forests: Financial and policy implications from the National Fire and Fire Surrogate Study. Forest Policy and Economics 10:344–354.

Haugo, R. D., B. S. Kellogg, C. A. Cansler, C. A. Kolden, K. B. Kemp, J. C. Robertson, K. L. Metlen, N. M. Vaillant, and C. M. Restaino. 2019. The missing fire: quantifying human exclusion of wildfire in Pacific Northwest forests, USA. Ecosphere 10:e02702.

Hessburg, P. F., et al. 2015. Restoring fire-prone Inland Pacific landscapes: seven core principles. Landscape Ecology 30:1805–1835.

Hessburg, P. F., et al. 2016. Tamm Review: Management of mixed-severity fire regime forests in Oregon, Washington, and Northern California. Forest Ecology and Management 366:221–250.

Hessburg, P. F., et al. 2019. Climate, environment, and disturbance history govern resilience of western North American forests. Frontiers in Ecology and Evolution 7:239.

Hessburg, P. F., et al. 2020. The 1994 eastside screens large-tree harvest limit: review of science relevant to forest planning 25 years later. General Technical Report PNW-GTR-990. U,.S. Forest Service, Portland, Oregon, USA.

Hessburg, P. F., J. K. Agee, and J. F. Franklin. 2005. Dry forests and wildland fires of the inland Northwest USA: Contrasting the landscape ecology of the pre-settlement and modern eras. Forest Ecology and Management 211:117–139.

Hessburg, P. F., S. J. Prichard, R. K. Hagmann, N. A. Povak, and F. K. Lake. 2021. Wildfire and climate change adaptation of western North American forests: a case for intentional management. Ecological Applications.

Hoffman, K. R., S. B. Wickham, W. S. McInnes, and B. M. Starzomski. 2019. Fire exclusion destroys habitats for at-risk species in a British Columbia protected area. Fire 2:48.

Holden, Z. A., P. Morgan, and A. T. Hudak. 2010. Burn severity of areas reburned by wildfires in the Gila National Forest, New Mexico, USA. Fire Ecology. 6:77–85.

Holden, Z. A., P. Morgan, M. G. Rollins, and K. Kavanagh. 2007. Effects of multiple wildland fires on ponderosa pine stand structure in two southwestern wilderness areas, USA. Fire Ecology 3:18–33.

Holl, K. D. 2020. Primer of ecological restoration. Island Press, Washington, D.C., USA.

Holl, K. D., and P. H. S. Brancalion. 2020. Tree planting is not a simple solution. Science 368:580–581.

Holsinger, L., S. A. Parks, and C. Miller. 2016. Weather, fuels, and topography impede wildland fire spread in western US landscapes. Forest Ecology and Management 380:59–69.

Hood, S., A. Sala, E. K. Heyerdahl, and M. Boutin. 2015. Low-severity fire increases tree defense against bark beetle attacks. Ecology 96:1846–1855.

Hudiburg, T., B. Law, D. P. Turner, J. Campbell, D. Donato, and M. Duane. 2009. Carbon dynamics of Oregon and northern California forests and potential land-based carbon storage. Ecological Applications 19:163–180.

Huffman, D. W., J. E. Crouse, A. J. Sánchez Meador, J. D. Springer, and M. T. Stoddard. 2018. Restoration benefits of re-entry with resource objective wildfire on a ponderosa pine landscape in northern Arizona, USA. Forest Ecology and Management 408:16–24.

Huffman, D. W., J. P. Roccaforte, J. D. Springer, and J. E. Crouse. 2020. Restoration applications of resource objective wildfires in western US forests: a status of knowledge review. Fire Ecology 16:18.

Huffman, D. W., A. J. Sánchez Meador, M. T. Stoddard, J. E. Crouse, and J. P. Roccaforte. 2017. Efficacy of resource objective wildfires for restoration of ponderosa pine (*Pinus ponderosa*) forests in northern Arizona. Forest Ecology and Management 389:395–403.

Huffman, M. 2013. The many elements of traditional fire knowledge: synthesis, classification, and aids to cross-cultural problem solving in fire-dependent systems around the world. Ecology and Society 18:3.

Hunter, M. E., J. M. Iniguez, and C. A. Farris. 2014. Historical and current fire management practices in two wilderness areas in the southwestern United States: the Saguaro Wilderness Area and the Gila-Aldo Leopold Wilderness Complex. General Technical Report RMRS-GTR-325. USDA Forest Service, Fort Collins, Colorado, USA.

Hurteau, M. D., G. W. Koch, and B. A. Hungate. 2008. Carbon protection and fire risk reduction: toward a full accounting of forest carbon offsets. Frontiers in Ecology and the Environment 6:493–498.

Hurteau, M. D., S. Liang, A. L. Westerling, and C. Wiedinmyer. 2019. Vegetation-fire feedback reduces projected area burned under climate change. Scientific Reports 9:2838.

Hurteau, M., and M. North. 2009. Fuel treatment effects on tree-based forest carbon storage and emissions under modeled wildfire scenarios. Frontiers in Ecology and the Environment 7:409–414.

Jain, T. B., J. S. Fried, and S. M. Loreno. 2020. Simulating the effectiveness of improvement cuts and commercial thinning to enhance fire resistance in west coast dry mixed conifer forests. Forest Science 66:157–177.

Jeronimo, S. M. A., V. R. Kane, D. J. Churchill, J. A. Lutz, M. P. North, G. P. Asner, and J. F. Franklin. 2019. Forest structure and pattern vary by climate and landform across active-fire landscapes in the montane Sierra Nevada. Forest Ecology and Management 437:70–86.

Johnson, M. C., M. C. Kennedy, S. C. Harrison, D. Churchill, J. Pass, and P. W. Fischer. 2020. Effects of post-fire management on dead woody fuel dynamics and stand structure in a severely burned mixed-conifer forest, in northeastern Washington State, USA. Forest Ecology and Management 470:118190.

Johnson, M. C., M. C. Kennedy, and D. L. Peterson. 2011. Simulating fuel treatment effects in dry forests of the western United States: testing the principles of a fire-safe forest. Canadian Journal of Forest Research 41:1018–1030.

Johnston, J. D., J. D. Bailey, and C. J. Dunn. 2016. Influence of fire disturbance and biophysical heterogeneity on pre-settlement ponderosa pine and mixed conifer forests. Ecosphere 7:e01581.

Johnstone, J. F., et al. 2016. Changing disturbance regimes, ecological memory, and forest resilience. Frontiers in Ecology and the Environment 14:369–378.
Jones, G. M., J. J. Keane, R. J. Gutiérrez, and M. Z. Peery. 2018. Declining old-forest species as a legacy of large trees lost. Diversity and Distributions 24:341–351.
Kalies, E. L., and L. Yocom Kent. 2016. Tamm Review: Are fuel treatments effective at achieving ecological and social objectives? A systematic review. Forest Ecology and Management 375:84–95.
Kane, J. M., J. M. Varner, and E. E. Knapp. 2009. Novel fuelbed characteristics associated with mechanical mastication treatments in northern California and south-western Oregon, USA. International Journal of Wildland Fire 18:686–697.
Kane, V. R., B. N. Bartl-Geller, M. P. North, J. T. Kane, J. M. Lydersen, S. M. Jeronimo, B. M. Collins, and L. M. Moskal. 2019. First-entry wildfires can create opening and tree clump patterns characteristic of resilient forests. Forest Ecology and Management 454:117659.
Kane, V. R., C. A. Cansler, N. A. Povak, J. T. Kane, R. J. McGaughey, J. A. Lutz, D. J. Churchill, and M. P. North. 2015. Mixed severity fire effects within the Rim fire: relative importance of local climate, fire weather, topography, and forest structure. Forest Ecology and Management 358:62–79.
Kane, V. R., M. P. North, J. A. Lutz, D. J. Churchill, S. L. Roberts, D. F. Smith, R. J. McGaughey, J. T. Kane, and M. L. Brooks. 2014. Assessing fire effects on forest spatial structure using a fusion of Landsat and airborne LiDAR data in Yosemite National Park. Remote Sensing of Environment 151:89–101.
Keane, R. E., S. Arno, and L. J. Dickinson. 2006. Wilderness management: the complexity of managing fire-dependent ecosystems in wilderness: relict ponderosa pine in the Bob Marshall Wilderness. Ecological Restoration 24:71–78.
Keane, R. E., P. F. Hessburg, P. B. Landres, and F. J. Swanson. 2009. The use of historical range and variability (HRV) in landscape management. Forest Ecology and Management 258:1025–1037.
Keane, R. E., D. McKenzie, D. A. Falk, E. A. H. Smithwick, C. Miller, and L.-K.-B. Kellogg. 2015. Representing climate, disturbance, and vegetation interactions in landscape models. Ecological Modelling 309–310:33–47.
Keeley, J. E., and A. D. Syphard. 2018. Historical patterns of wildfire ignition sources in California ecosystems. International Journal of Wildland Fire 27:781–799.
Kemp, K. B., P. E. Higuera, P. Morgan, and J. T. Abatzoglou. 2019. Climate will increasingly determine post-fire tree regeneration success in low-elevation forests, Northern Rockies, USA. Ecosphere 10:e02568.
Kennedy, M. C., and M. C. Johnson. 2014. Fuel treatment prescriptions alter spatial patterns of fire severity around the wildland–urban interface during the Wallow Fire, Arizona, USA. Forest Ecology and Management 318:122–132.
Kennedy, M. C., M. C. Johnson, K. Fallon, and D. Mayer. 2019. How big is enough? Vegetation structure impacts effective fuel treatment width and forest resiliency. Ecosphere 10: e02573.
Kimmerer, R. W., and F. K. Lake. 2001. The role of indigenous burning in land management. Journal of Forestry 11:36–41.
Knapp, E. E., J. M. Lydersen, M. P. North, and B. M. Collins. 2017. Efficacy of variable density thinning and prescribed fire for restoring forest heterogeneity to mixed-conifer forest in the central Sierra Nevada, CA. Forest Ecology and Management 406:228–241.
Knapp, E. E., C. N. Skinner, M. P. North, and B. L. Estes. 2013. Long-term overstory and understory change following logging and fire exclusion in a Sierra Nevada mixed-conifer forest. Forest Ecology and Management 310:903–914.
Kobziar, L. N., J. R. McBride, and S. L. Stephens. 2009. The efficacy of fire and fuels reduction treatments in a Sierra Nevada pine plantation. International Journal of Wildland Fire 18:791–801.
Kolden, C. A. 2019. We're not doing enough prescribed fire in the western United States to mitigate wildfire risk. Fire 2:30.
Kolden, C. A., and T. J. Brown. 2010. Beyond wildfire: perspectives of climate, managed fire and policy in the USA. International Journal of Wildland Fire 19:364–373.
Kolden, C. A., and C. Henson. 2019. A socio-ecological approach to mitigating wildfire vulnerability in the wildland urban interface: a case study from the 2017 Thomas Fire. Fire 2:9.
Koontz, M. J., M. P. North, C. M. Werner, S. E. Fick, and A. M. Latimer. 2020. Local forest structure variability increases resilience to wildfire in dry western U.S. coniferous forests. Ecology Letters 23:483–494.
Korb, J. E., P. J. Fornwalt, and C. S. Stevens-Rumann. 2019. What drives ponderosa pine regeneration following wildfire in the western United States? Forest Ecology and Management 454:117663.
Korb, J. E., M. T. Stoddard, and D. W. Huffman. 2020. Effectiveness of restoration treatments for reducing fuels and increasing understory diversity in shrubby mixed-conifer forests of the southern Rocky Mountains, USA. Forests 11:508.
Krawchuk, M. A., G. W. Meigs, J. M. Cartwright, J. D. Coop, R. Davis, A. Holz, C. Kolden, and A. J. Meddens. 2020. Disturbance refugia within mosaics of forest fire, drought, and insect outbreaks. Frontiers in Ecology and the Environment 18:235–244.
Kreye, J. K., N. W. Brewer, P. Morgan, J. M. Varner, A. M. S. Smith, C. M. Hoffman, and R. D. Ottmar. 2014. Fire behavior in masticated fuels: a review. Forest Ecology and Management 314:193–207.
Krofcheck, D. J., M. D. Hurteau, R. M. Scheller, and E. L. Loudermilk. 2017. Restoring surface fire stabilizes forest carbon under extreme fire weather in the Sierra Nevada. Ecosphere 8:e01663.
Krofcheck, D. J., C. C. Remy, A. R. Keyser, and M. D. Hurteau. 2019. Optimizing forest management stabilizes carbon under projected climate and wildfires. Journal of Geophysical Research: Biogeosciences 124:3075–3087.
Lake, F. K., and A. C. Christianson. 2019. Indigenous fire stewardship. Pages 1–9 *in* S. L. Manzello, editor. Encyclopedia of Wildfires and Wildland-Urban Interface (WUI) Fires. Springer Nature, Cham, Switzerland.
Lake, F. K., J. Parrotta, C. P. Giardina, I. Davidson-Hunt, and Y. Uprety. 2018. Chapter 12: Integration of traditional and western knowledge in forest landscape restoration. *In* S. Mansourian and J. Parrotta, editors. Forest Landscape Restoration: integrated approaches to support effective implementation. Routledge, Oxfordshire, UK.
Lake, F. K., V. Wright, P. Morgan, M. McFadzen, D. McWethy, and C. S. Stevens-Rumann. 2017. Returning fire to the land: celebrating traditional knowledge and fire. Journal of Forestry 115:343–353.
Landis, T. D., L. E. Riley, D. L. Haase, and J. R. Pinto. 2011. The target plant concept-a history and brief overview. National Proceedings: Forest and Conservation Nursery Associations-2010 Proceedings. RMRS-P-65. USDA Forest Service, Fort Collins, Colorado, USA.
Larson, A. J., R. T. Belote, C. A. Cansler, S. A. Parks, and M. S. Dietz. 2013*a*. Latent resilience in ponderosa pine forest: effects of resumed frequent fire. Ecological Applications 23:1243–1249.
Larson, A. J., R. T. Belote, M. A. Williamson, and G. H. Aplet. 2013*b*. Making monitoring count: project design for active adaptive management. Journal of Forestry 5:348–356.

Larson, A. J., and D. Churchill. 2012. Tree spatial patterns in fire-frequent forests of western North America, including mechanisms of pattern formation and implications for designing fuel reduction and restoration treatments. Forest Ecology and Management 267:74–92.

LeFevre, M. E., D. J. Churchill, A. J. Larson, S. M. A. Jeronimo, J. Bass, J. F. Franklin, and V. R. Kane. 2020. Evaluating restoration treatment effectiveness through a comparison of residual composition, structure, and spatial pattern with historical reference sites. Forest Science 66:578–588.

Lehmkuhl, J., W. Gaines, D. Peterson, J. Bailey, and A. Youngblood. 2015. Silviculture and monitoring guidelines for integrating restoration of dry mixed-conifer forest and spotted owl habitat management in the eastern Cascade Range. General Technical Report PNW-GTR-915. USDA Forest Service, Portland, Oregon, USA.

Lehmkuhl, J. F., M. Kennedy, E. D. Ford, P. H. Singleton, W. L. Gaines, and R. L. Lind. 2007. Seeing the forest for the fuel: integrating ecological values and fuels management. Forest Ecology and Management 246:73–80.

Levine, C. R., C. V. Cogbill, B. M. Collins, A. J. Larson, J. A. Lutz, M. P. North, C. M. Restaino, H. D. Safford, S. L. Stephens, and J. J. Battles. 2017. Evaluating a new method for reconstructing forest conditions from General Land Office survey records. Ecological Applications 27:1498–1513.

Liang, S., M. D. Hurteau, and A. L. Westerling. 2017. Potential decline in carbon carrying capacity under projected climate-wildfire interactions in the Sierra Nevada. Scientific Reports 7:2420.

Liang, S., M. D. Hurteau, and A. L. Westerling. 2018. Large-scale restoration increases carbon stability under projected climate and wildfire regimes. Frontiers in Ecology and the Environment 16:207–212.

Lilley, P. L., and M. Vellend. 2009. Negative native-exotic diversity relationship in oak savannas explained by human influence and climate. Oikos 118:1373–1382.

Littell, J. S., D. McKenzie, H. Y. Wan, and S. A. Cushman. 2018. Climate change and future wildfire in the western United States: an ecological approach to nonstationarity. Earth's Future 6:1097–1111.

Long, J. W., and F. K. Lake. 2018. Escaping social-ecological traps through tribal stewardship on national forest lands in the Pacific Northwest, United States of America. Ecology and Society 23:10.

Lutz, J. A., et al. 2020. The importance of large-diameter trees to fuel evolution following reintroduced fire in a mixed-conifer forest in Yosemite National Park, California, USA. Ecological Processes 9:41.

Lydersen, J. M., B. M. Collins, M. Coppoletta, M. R. Jaffe, H. Northrop, and S. L. Stephens. 2019*a*. Fuel dynamics and reburn severity following high-severity fire in a Sierra Nevada, USA, mixed-conifer forest. Fire Ecology 15:43.

Lydersen, J. M., B. M. Collins, and C. T. Hunsaker. 2019*b*. Implementation constraints limit benefits of restoration treatments in mixed-conifer forests. International Journal of Wildland Fire 28:495–511.

Lydersen, J. M., B. M. Collins, J. D. Miller, D. L. Fry, and S. L. Stephens. 2016. Relating fire-caused change in forest structure to remotely sensed estimates of fire severity. Fire Ecology 12:99–116.

Lydersen, J., M. North, and B. M. Collins. 2014. Severity of an uncharacteristically large wildfire, the Rim Fire, in forests with relatively restored frequent fire regimes. Forest Ecology and Management 328:326–334.

Lyons, J. E., M. C. Runge, H. P. Laskowski, and W. L. Kendall. 2008. Monitoring in the context of structured decision-making and adaptive management. Journal of Wildlife Management 72:1683–1692.

Mallek, C., H. Safford, J. Viers, and J. Miller. 2013. Modern departures in fire severity and area vary by forest type, Sierra Nevada and southern Cascades, California, USA. Ecosphere 4:1–28.

Margolis, E. Q., and S. B. Malevich. 2016. Historical dominance of low-severity fire in dry and wet mixed-conifer forest habitats of the endangered terrestrial Jemez Mountains salamander (*Plethodon neomexicanus*). Forest Ecology and Management 375:12–26.

Marks-Block, T., F. K. Lake, and L. M. Curran. 2019. Effects of understory fire management treatments on California hazelnut, an ecocultural resource of the Karuk and Yurok Indians in the Pacific Northwest. Forest Ecology and Management 450:117517.

Marlon, J. R., et al. 2012. Long-term perspective on wildfires in the western USA. Proceedings of the National Academy of Sciences USA 109:535–543.

Martinson, E. J., and P. N. Omi. 2013. Fuel treatments and fire severity: a meta-analysis. Research Paper RMRS-RP-103www. USDA Forest Service, Fort Collins, Colorado, USA.

Martinson, E., P. N. Omi, and W. Shepperd. 2003. Part 3: Effects of fuel treatments on fire severity. Pages 96–126 *in* R. T. Graham, editor. Hayman fire case study. General Technical Report RMRS-GTR-114. USDA Forest Service, Ogden, Utah, USA.

McCarley, T. R., C. A. Kolden, N. M. Vaillant, A. T. Hudak, A. M. Smith, and J. Kreitler. 2017. Landscape-scale quantification of fire-induced change in canopy cover following mountain pine beetle outbreak and timber harvest. Forest Ecology and Management 391:164–175.

McKenzie, D., and J. S. Littell. 2017. Climate change and the eco-hydrology of fire: Will area burned increase in a warming western USA? Ecological Applications 27:26–36.

McWethy, D. B., et al. 2019. Rethinking resilience to wildfire. Nature Sustainability 2:797–804.

Meddens, A. J. H., C. A. Kolden, J. A. Lutz, A. M. S. Smith, C. A. Cansler, J. T. Abatzoglou, G. W. Meigs, W. M. Downing, and M. A. Krawchuk. 2018. Fire refugia: what are they, and why do they matter for global change? BioScience 68:944–954.

Mell, W. E., S. L. Manzello, A. Maranghides, D. Butry, and R. G. Rehm. 2010. The wildland–urban interface fire problem—current approaches and research needs. International Journal of Wildland Fire 19:238–251.

Merschel, A. G., E. K. Heyerdahl, T. A. Spies, and R. A. Loehman. 2018. Influence of landscape structure, topography, and forest type on spatial variation in historical fire regimes, Central Oregon, USA. Landscape Ecology 33:1195–1209.

Merschel, A. G., T. A. Spies, and E. K. Heyerdahl. 2014. Mixed-conifer forests of central Oregon: effects of logging and fire exclusion vary with environment. Ecological Applications 24:1670–1688.

Meyer, M. D. 2015. Forest fire severity patterns of resource objective wildfires in the southern Sierra Nevada. Journal of Forestry 113:49–56.

Miller, J. D., and B. Quayle. 2015. Calibration and validation of immediate post-fire satellite-derived data to three severity metrics. Fire Ecology 11:12–30.

Miller, J. D., and H. Safford. 2012. Trends in wildfire severity: 1984 to 2010 in the Sierra Nevada, Modoc Plateau, and southern Cascades, California, USA. Fire Ecology 8:41–57.

Miller, R. K., C. B. Field, and K. J. Mach. 2020. Barriers and enablers for prescribed burns for wildfire management in California. Nature Sustainability 3:101–109.

Moghaddas, J. J., and L. Craggs. 2007. A fuel treatment reduces fire severity and increases suppression efficiency in a mixed conifer forest. International Journal of Wildland Fire 16:673.

Moreira, F., et al. 2020. Wildfire management in Mediterranean-type regions: paradigm change needed. Environmental Research Letters 15:1.

Morgan, P., E. K. Heyerdahl, and C. E. Gibson. 2008. Multi-season climate synchronized forest fires throughout the 20th century, northern Rockies, USA. Ecology 89:717–728.

Morgan, P., A. T. Hudak, A. Wells, S. A. Parks, L. S. Baggett, B. C. Bright, and P. Green. 2017. Multidecadal trends in area burned with high severity in the Selway-Bitterroot Wilderness Area 1880–2012. International Journal of Wildland Fire 26:930–943.

Moritz, M. A., et al. 2014. Learning to coexist with wildfire. Nature 515:58–66.

Naficy, C. E., E. G. Keeling, P. Landres, P. F. Hessburg, T. T. Veblen, and A. Sala. 2016. Wilderness in the 21st century: a framework for testing assumptions about ecological intervention in wilderness using a case study of fire ecology in the Rocky Mountains. Journal of Forestry 114:384–395.

Ng, J., M. P. North, A. J. Arditti, M. R. Cooper, and J. A. Lutz. 2020. Topographic variation in tree group and gap structure in Sierra Nevada mixed-conifer forests with active fire regimes. Forest Ecology and Management 472:118220.

Nigro, K., and N. Molinari. 2019. Status and trends of fire activity in southern California yellow pine and mixed conifer forests. Forest Ecology and Management 441:20–31.

North, M. P., et al. 2019. Tamm Review: Reforestation for resilience in dry western U.S. forests. Forest Ecology and Management 432:209–224.

North, M., A. Brough, J. Long, B. Collins, P. Bowden, D. Yasuda, J. Miller, and N. Sugihara. 2015*a*. Constraints on mechanized treatment significantly limit mechanical fuels reduction extent in the Sierra Nevada. Journal of Forestry 113:40–48.

North, M., B. M. Collins, and S. Stephens. 2012. Using fire to increase the scale, benefits, and future maintenance of fuels treatments. Journal of Forestry 110:392–401.

North, M. P., S. L. Stephens, B. M. Collins, J. K. Agee, G. Aplet, J. F. Franklin, and P. Z. Fulé. 2015*b*. Reform forest fire management. Science 349:1280–1281.

North, M., P. Stine, K. O'Hara, W. Zielinski, and S. Stephens. 2009. An ecosystem management strategy for Sierran mixed-conifer forests. General Technical Report PSW-GTR-220. USDA Forest Service, Albany, California, USA.

Odion, D. C., et al. 2014. Examining historical and current mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. PLoS One 9:e87852.

Odion, D. C., and C. T. Hanson. 2006. Fire severity in conifer forests of the Sierra Nevada, California. Ecosystems 9:1177–1189.

Parisien, M.-A., Q. E. Barber, K. G. Hirsch, C. A. Stockdale, S. Erni, X. Wang, D. Arseneault, and S. A. Parks. 2020. Fire deficit increases wildfire risk for many communities in the Canadian boreal forest. Nature Communications 11:2121.

Parisien, M.-A., and M. A. Moritz. 2009. Environmental controls on the distribution of wildfire at multiple spatial scales. Ecological Monographs 79:127–154.

Parisien, M.-A., S. Snetsinger, J. A. Greenberg, C. R. Nelson, T. Schoennagel, S. Z. Dobrowski, and M. A. Moritz. 2012. Spatial variability in wildfire probability across the western United States. International Journal of Wildland Fire 21:313–327.

Park, J., G. Walker, W. Hunt, D. Bourdin, and J. Weir. 2019. Banff, Kootenay and Yoho National Parks Fire management plan. Parks Canada, Calgary, Alberta, Canada.

Parks, S. A. 2014. Mapping day-of-burning with coarse-resolution satellite fire-detection data. International Journal of Wildland Fire 23:215–223.

Parks, S. A., and J. T. Abatzoglou. 2020. Warmer and drier fire seasons contribute to increases in area burned at high severity in western US forests from 1985–2017. Geophysical Research Letters 47:e2020GL089858.

Parks, S., S. Dobrowski, and M. Panunto. 2018. What drives low-severity fire in the southwestern USA? Forests 9:165.

Parks, S. A., S. Z. Dobrowski, J. D. Shaw, and C. Miller. 2019. Living on the edge: trailing edge forests at risk of fire-facilitated conversion to nonforest. Ecosphere 10:e02651.

Parks, S. A., L. M. Holsinger, C. Miller, and C. R. Nelson. 2015*a*. Wildland fire as a self-regulating mechanism: the role of previous burns and weather in limiting fire progression. Ecological Applications 25:1478–1492.

Parks, S. A., C. Miller, L. M. Holsinger, L. S. Baggett, and B. J. Bird. 2016. Wildland fire limits subsequent fire occurrence. International Journal of Wildland Fire 25:182–190.

Parks, S. A., C. Miller, C. R. Nelson, and Z. A. Holden. 2014. Previous fires moderate burn severity of subsequent wildland fires in two large western US wilderness areas. Ecosystems 17:29–42.

Parks, S. A., C. Miller, M.-A. Parisien, L. M. Holsinger, S. Z. Dobrowski, and J. Abatzoglou. 2015*b*. Wildland fire deficit and surplus in the western United States, 1984–2012. Ecosphere 6:275.

Parsons, R. A., F. Pimont, L. Wells, G. Cohn, W. M. Jolly, F. de Coligny, E. Rigolot, J.-L. Dupuy, W. Mell, and R. R. Linn. 2018. Modeling thinning effects on fire behavior with STANDFIRE. Annals of Forest Science 75:7.

Perry, D. A., P. F. Hessburg, C. N. Skinner, T. A. Spies, S. L. Stephens, A. H. Taylor, J. F. Franklin, B. McComb, and G. Riegel. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and northern California. Forest Ecology and Management 262:703–717.

Peterson, D. W., E. K. Dodson, and R. J. Harrod. 2015. Post-fire logging reduces surface woody fuels up to four decades following wildfire. Forest Ecology and Management 338:84–91.

Peterson, D. W., P. F. Hessburg, R. B. Salter, K. M. James, M. C. Dahlgreen, and J. A. Barnes. 2007. Reintroducing fire in regenerated dry forests following stand-replacing wildfire. General Technical Report PSW-GTR-203. USDA Forest Service, Albany, California, USA.

Pimont, F., J.-L. Dupuy, R. R. Linn, and S. Dupont. 2009. Validation of FIRETEC wind-flows over a canopy and a fuel-break. International Journal of Wildland Fire 18:775–790.

Pollet, J., and P. N. Omi. 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. International Journal of Wildland Fire 11:1–10.

Povak, N. A., P. F. Hessburg, and R. B. Salter. 2018. Evidence for scale-dependent topographic controls on wildfire spread. Ecosphere 9:e02443.

Povak, N. A., V. R. Kane, B. M. Collins, J. M. Lydersen, and J. T. Kane. 2020. Multi-scaled drivers of severity patterns vary across land ownerships for the 2013 Rim Fire, California. Landscape Ecology 35:293–318.

Power, M. J., B. F. Codding, A. H. Taylor, T. W. Swetnam, K. E. Magargal, D. W. Bird, and J. F. O'Connell. 2018. Human fire legacies on ecological landscapes. Frontiers in Earth Science 6:151.

Prichard, S. J., and M. C. Kennedy. 2012. Fuel treatment effects on tree mortality following wildfire in dry mixed conifer forests, Washington State, USA. International Journal of Wildland Fire 21:1004–1013.

Prichard, S. J., and M. C. Kennedy. 2014. Fuel treatments and landform modify landscape patterns of burn severity in an extreme fire event. Ecological Applications. 24:571–590.

Prichard, S. J., D. L. Peterson, and K. Jacobson. 2010. Fuel treatments reduce the severity of wildfire effects in dry mixed conifer forest, Washington, USA. Canadian Journal of Forest Research 40:1615–1626.

Prichard, S. J., N. A. Povak, M. C. Kennedy, and D. W. Peterson. 2020. Fuel treatment effectiveness in the context of landform, vegetation, and large, wind-driven wildfires. Ecological Applications 30:e02104.

Prichard, S. J., C. S. Stevens-Rumann, and P. F. Hessburg. 2017. Tamm Review: Shifting global fire regimes: Lessons from reburns and research needs. Forest Ecology and Management 396:217–233.

Quinn-Davidson, L. N., and J. M. Varner. 2012. Impediments to prescribed fire across agency, landscape and manager: an example from northern California. International Journal of Wildland Fire 21:210–218.

Radeloff, V. C., et al. 2018. Rapid growth of the US wildland-urban interface raises wildfire risk. Proceedings of the National Academy of Sciences USA 115:3314–3319.

Reilly, M. J., C. J. Dunn, G. W. Meigs, T. A. Spies, R. E. Kennedy, J. D. Bailey, and K. Briggs. 2017. Contemporary patterns of fire extent and severity in forests of the Pacific Northwest, USA (1985–2010). Ecosphere 8:e01695.

Reilly, M. J., M. Elia, T. A. Spies, M. J. Gregory, G. Sanesi, and R. Lafortezza. 2018. Cumulative effects of wildfires on forest dynamics in the eastern Cascade Mountains, USA. Ecological Applications 28:291–308.

Reinhardt, E. D., R. E. Keane, D. E. Calkin, and J. D. Cohen. 2008. Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. Forest Ecology and Management 256:1997–2006.

Reinhardt, E., J. Scott, K. Gray, and R. Keane. 2006. Estimating canopy fuel characteristics in five conifer stands in the western United States using tree and stand measurements. Canadian Journal of Forest Research 36:2803–2814.

Restaino, J. C., and D. L. Peterson. 2013. Wildfire and fuel treatment effects on forest carbon dynamics in the western United States. Forest Ecology and Management 303:46–60.

Reynolds, R. T., A. J. S. Meador, J. A. Youtz, T. Nicolet, M. S. Matonis, P. L. Jackson, D. G. DeLorenzo, and A. D. Graves. 2013. Restoring composition and structure in Southwestern frequent-fire forests: A science-based framework for improving ecosystem resiliency. General Technical Report RMRS-GTR-310. USDA Forest Service, Fort Collins, Colorado, USA.

Rhodes, J. J., and W. L. Baker. 2008. Fire probability, fuel treatment effectiveness and ecological tradeoffs in western U.S. public forests. Open Forest Science Journal 1:1–7.

Ritchie, M. W., E. E. Knapp, and C. N. Skinner. 2013. Snag longevity and surface fuel accumulation following post-fire logging in a ponderosa pine dominated forest. Forest Ecology and Management 287:113–122.

Ritter, S. M., C. M. Hoffman, M. A. Battaglia, C. S. Stevens-Rumann, and W. E. Mell. 2020. Fine-scale fire patterns mediate forest structure in frequent-fire ecosystems. Ecosphere 11: e03177.

Rivera-Huerta, H., H. D. Safford, and J. D. Miller. 2016. Patterns and trends in burned area and fire severity from 1984 to 2010 in the Sierra de San Pedro Mártir, Baja California, Mexico. Fire Ecology 12:52–72.

Rodman, K. C., T. T. Veblen, T. B. Chapman, M. T. Rother, A. P. Wion, and M. D. Redmond. 2020. Limitations to recovery following wildfire in dry forests of southern Colorado and northern New Mexico, USA. Ecological Applications 30: e02001.

Rollins, M. G., P. Morgan, and T. Swetnam. 2002. Landscape-scale controls over 20th century fire occurrence in two large Rocky Mountain (USA) wilderness areas. Landscape Ecology 17:539–557.

Roos, C. L., et al. 2021. Native American fire management at an ancient wildland–urban interface in the Southwest United States. Proceedings of the National Academy of Sciences USA 118:e2018733118.

Rowe, J. S., and G. W. Scotter. 1973. Fire in the boreal forest. Quaternary Research 3:444–464.

Ryan, K. C., E. E. Knapp, and M. J. Varner. 2013. Prescribed fire in North American forests and woodlands: history, current practice, and challenges. Frontiers in Ecology and the Environment 11:15–24.

Safford, H. D., G. D. Hayward, N. E. Heller, and J. Wiens. 2012*a*. Historical ecology, climate change, and resource management: can the past still inform the future? Pages 46–62 *in* J. A. Wiens, G. D. Hayward, H. D. Safford, and C. M. Giffen, editors. Historical environmental variation in conservation and natural resource management. John Wiley & Sons, Hoboken, New Jersey, USA.

Safford, H. D., D. A. Schmidt, and C. H. Carlson. 2009. Effects of fuel treatments on fire severity in an area of wildland–urban interface, Angora Fire, Lake Tahoe Basin, California. Forest Ecology and Management 258:773–787.

Safford, H. D., and J. T. Stevens. 2017. Natural Range of Variation (NRV) for yellow pine and mixed conifer forests in the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests, California, USA. General Technical Report PSW-GTR-256. USDA Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Safford, H. D., J. T. Stevens, K. Merriam, M. D. Meyer, and A. M. Latimer. 2012*b*. Fuel treatment effectiveness in California yellow pine and mixed conifer forests. Forest Ecology and Management 274:17–28.

Savage, M., J. N. Mast, and J. J. Feddema. 2013. Double whammy: high-severity fire and drought in ponderosa pine forests of the Southwest. Canadian Journal of Forest Research 43:570–583.

Schmidt, D. A., A. H. Taylor, and C. N. Skinner. 2008. The influence of fuels treatment and landscape arrangement on simulated fire behavior, southern Cascade Range, California. Forest Ecology and Management 255:3170–3184.

Schoennagel, T., et al. 2017. Adapt to more wildfire in western North American forests as climate changes. Proceedings of the National Academy of Sciences USA 114:4582–4590.

Schoennagel, T., C. R. Nelson, D. M. Theobald, G. C. Carnwath, and T. B. Chapman. 2009. Implementation of National Fire Plan treatments near the wildland-urban interface in the western United States. Proceedings of the National Academy of Sciences USA 106:10706–10711.

Schoennagel, T., T. T. Veblen, and W. H. Romme. 2004. The interaction of fire, fuels, and climate across rocky mountain forests. BioScience 54:661–676.

Scholl, A. E., and A. H. Taylor. 2010. Fire regimes, forest change, and self-organization in an old-growth mixed-conifer forest, Yosemite National Park, USA. Ecological Applications 20:362–380.

Schultz, C. A., and T. Jedd. 2012. The collaborative forest landscape restoration program: a history and overview of the first projects. Journal of Forestry 110:381–391.

Schultz, C. A., S. M. McCaffrey, and H. R. Huber-Stearns. 2019. Policy barriers and opportunities for prescribed fire application in the western United States. International Journal of Wildland Fire 28:874.

Shive, K. L., H. K. Preisler, K. R. Welch, H. D. Safford, R. J. Butz, K. L. O'Hara, and S. L. Stephens. 2018. From the stand scale to the landscape scale: predicting the spatial patterns of forest regeneration after disturbance. Ecological Applications 28:1626–1639.

Smith, A. M. S., et al. 2016. The science of firescapes: achieving fire-resilient communities. BioScience 66:130–146.

Sneeuwjagt, R. J., T. S. Kline, and S. L. Stephens. 2013. Opportunities for improved fire use and management in California: lessons from western Australia. Fire Ecology 9:14–25.

Sowerwine, J., D. Sarna-Wojcicki, M. Mucioki, L. Hillman, F. K. Lake, and E. Friedman. 2019. Enhancing Indigenous food sovereignty: A five-year collaborative tribal-university research and extension project in California and Oregon. Journal of Agriculture, Food Systems, and Community Development. 9:167–190.

Spies, T. A., P. A. Stine, R. A. Gravenmier, J. W. Long, and M. J. Reilly. 2018. Volume 1—Synthesis of science to inform land management within the Northwest Forest Plan Area. General Technical Report PNW-GTR-966. USDA Forest Service, Portland, Oregon, USA.

Stan, A. B., P. Z. Fulé, K. B. Ireland, and J. S. Sanderlin. 2014. Modern fire regime resembles historical fire regime in a ponderosa pine forest on Native American lands. International Journal of Wildland Fire 23:686–697.

Stenzel, J. E., et al. 2019. Fixing a snag in carbon emissions estimates from wildfires. Global Change Biology 25:3985–3994.

Stephens, S. L., et al. 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western US forests. Ecological Applications 19:305–320.

Stephens, S. L., et al. 2021. Forest restoration and fuels reduction: convergent or divergent? BioScience 71:85–101.

Stephens, S. L., B. M. Collins, E. Biber, and P. Z. Fulé. 2016. U.S. federal fire and forest policy: emphasizing resilience in dry forests. Ecosphere 7:1–19.

Stephens, S. L., and P. Z. Fulé. 2005. Western pine forests with continuing frequent fire regimes: Possible reference sites for management. Journal of Forestry 103:357–362.

Stephens, S. L., J. D. McIver, R. E. J. Boerner, C. J. Fettig, J. B. Fontaine, B. R. Hartsough, P. L. Kennedy, and D. W. Schwilk. 2012. The effects of forest fuel-reduction treatments in the United States. BioScience 62:549–560.

Stephens, S. L., C. I. Millar, and B. M. Collins. 2010. Operational approaches to managing forests of the future in Mediterranean regions within a context of changing climates. Environmental Research Letters 5:024003.

Stephens, S. L., and J. J. Moghaddas. 2005. Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. Forest Ecology and Management 215:21–36.

Stephens, S. L., A. L. Westerling, M. D. Hurteau, Z. M. Peery, C. A. Schultz, and S. Thompson. 2020. Fire and climate change: conserving seasonally dry forests is still possible. Frontiers in Ecology and the Environment 18:354–360.

Stevens, J. T., B. M. Collins, J. W. Long, M. P. North, S. J. Prichard, L. W. Tarnay, and A. M. White. 2016. Evaluating potential trade-offs among fuel treatment strategies in mixed-conifer forests of the Sierra Nevada. Ecosphere 7:e01445.

Stevens, J. T., B. M. Collins, J. D. Miller, M. P. North, and S. L. Stephens. 2017. Changing spatial patterns of stand-replacing fire in California conifer forests. Forest Ecology and Management 406:28–36.

Stevens, J. T., M. M. Kling, D. W. Schwilk, J. M. Varner, and J. M. Kane. 2020. Biogeography of fire regimes in western U.S. conifer forests: A trait-based approach. Global Ecology and Biogeography 29:944–955.

Stevens, J. T., H. D. Safford, and A. M. Latimer. 2014. Wildfire-contingent effects of fuel treatments can promote ecological resilience in seasonally dry conifer forests. Canadian Journal of Forest Research 44:843–854.

Stevens-Rumann, C. S., and P. Morgan. 2019. Tree regeneration following wildfires in the western US: a review. Fire Ecology 15:15.

Stevens-Rumann, C. S., S. J. Prichard, E. K. Strand, and P. Morgan. 2016. Prior wildfires influence burn severity of subsequent large fires. Canadian Journal of Forest Research 46:1375–1385.

Stevens-Rumann, C. S., C. H. Sieg, and M. E. Hunter. 2012. Ten years after wildfires: How does varying tree mortality impact fire hazard and forest resiliency? Forest Ecology and Management 267:199–208.

Stine, P., et al. 2014. The ecology and management of moist mixed-conifer forests in eastern Oregon and Washington: a synthesis of the relevant biophysical science and implications for future land management. General Technical Report PNW-GTR-897. USDA Forest Service, Portland, Oregon, USA.

Stockdale, C. A., S. E. MacDonald, and E. Higgs. 2019. Forest closure and encroachment at the grassland interface: a century-scale analysis using oblique repeat photography. Ecosphere 10:e02774.

Stoddard, M. T., P. Z. Fulé, D. W. Huffman, A. J. S. Meador, and J. P. Roccaforte. 2020. Ecosystem management applications of resource objective wildfires in forests of the Grand Canyon National Park, USA. International Journal of Wildland Fire 29:190–200.

Storm, L., and D. Shebitz. 2006. Evaluating the purpose, extent, and ecological restoration applications of indigenous burning practices in southwestern Washington. Ecological Restoration 24:256–268.

Striplin, R., S. A. McAfee, H. D. Safford, and M. J. Papa. 2020. Retrospective analysis of burn windows for fire and fuels management: an example from the Lake Tahoe Basin, California, USA. Fire Ecology 16:13.

Syphard, A. D., J. E. Keeley, A. B. Massada, T. J. Brennan, and V. C. Radeloff. 2012. Housing arrangement and location determine the likelihood of housing loss due to wildfire. PLoS One 7:e33954.

Taylor, A. H., V. Trouet, C. N. Skinner, and S. Stephens. 2016. Socio-ecological transitions trigger fire regime shifts and modulate fire-climate interactions in the Sierra Nevada, CA, 1600–2015 CE. Proceedings of the National Academy of Sciences USA 13:13684–13689.

Teske, C. 2012. Characterizing fire-on-fire interactions in three large wilderness areas. Fire Ecology 8:82–106.

Thode, A. E., J. W. van Wagtendonk, J. D. Miller, and J. F. Quinn. 2011. Quantifying the fire regime distributions for severity in Yosemite National Park, California, USA. International Journal of Wildland Fire 20:223–239.

Thompson, D. K., M.-A. Parisien, J. Morin, K. Millard, C. P. S. Larsen, and B. N. Simpson. 2017. Fuel accumulation in a high-frequency boreal wildfire regime: from wetland to upland. Canadian Journal of Forest Research 47:957–964.

Thompson, J. R., T. A. Spies, and L. M. Ganio. 2007. Reburn severity in managed and unmanaged vegetation in a large wildfire. Proceedings of the National Academy of Sciences USA 104:10743–10748.

Thompson, M. P., P. Bowden, A. Brough, J. H. Scott, J. Gilbertson-Day, A. Taylor, J. Anderson, and J. R. Haas. 2016. Application of wildfire risk assessment results to wildfire response planning in the southern Sierra Nevada, California, USA. Forests 7:64.

Trauernicht, C., B. W. Brook, B. P. Murphy, G. J. Williamson, and D. M. J. S. Bowman. 2015. Local and global pyrogeographic evidence that indigenous fire management creates pyrodiversity. Ecology and Evolution 5:1908–1918.

Turner, M. G., K. H. Braziunas, W. D. Hansen, and B. J. Harvey. 2019. Short-interval severe fire erodes the resilience of subalpine lodgepole pine forests. Proceedings of the National Academy of Sciences USA 116:11319–11328.

Turner, M. G., and W. H. Romme. 1994. Landscape dynamics in crown fire ecosystems. Landscape Ecology 9:59–77.

Tymstra, C., B. J. Stocks, X. Cai, and M. D. Flannigan. 2020. Wildfire management in Canada: Review, challenges and opportunities. Progress in Disaster Science 5:100045.

Underwood, S., L. Arguello, and N. Siefkin. 2003. Restoring ethnographic landscapes and natural elements in Redwood National Park. Ecological Restoration 21:278–283.

United States Department of Agricuture and United States Department of Interior. 2021. National Cohesive Wildland Fire Management Strategy. https://www.forestsandrangelands.gov/strategy

United States Department of Agriculture. 2001. National fire plan research and development 2001 business summary. North Central Research Station, Saint Paul, Minnesota, USA.

Vaillant, N. M., J. A. Fites-Kaufman, and S. L. Stephens. 2009. Effectiveness of prescribed fire as a fuel treatment in Californian coniferous forests. International Journal of Wildland Fire 18:165–175.

Vaillant, N. M., and E. D. Reinhardt. 2017. An evaluation of the forest service hazardous fuels treatment program—Are we treating enough to promote resiliency or reduce hazard? Journal of Forestry 115:300–308.

Van de Water, K., and M. North. 2011. Stand structure, fuel loads, and fire behavior in riparian and upland forests, Sierra Nevada Mountains, USA: a comparison of current and reconstructed conditions. Forest Ecology and Management 262:215–228.

van Mantgem, P. J., D. A. Falk, E. C. Williams, A. J. Das, and N. L. Stephenson. 2018. Pre-fire drought and competition mediate post-fire conifer mortality in western U.S. National Parks. Ecological Applications 28:1730–1739.

van Wagtendonk, J. W., K. A. van Wagtendonk, and A. E. Thode. 2012. Factors associated with the severity of intersecting fires in Yosemite National Park, California, USA. Fire Ecology 7:11–31.

Veldman, J. W., et al. 2019. Comment on “The global tree restoration potential”. Science 366:eaay7976.

Walker, R. B., J. D. Coop, S. A. Parks, and L. Trader. 2018. Fire regimes approaching historic norms reduce wildfire-facilitated conversion from forest to nonforest. Ecosphere 9: e02182.

Westerling, A. L. 2016. Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. Philosophical Transactions of the Royal Society B 371:20150178.

Westlind, D. J., and B. K. Kerns. 2021. Repeated fall prescribed fire in previously thinned *Pinus ponderosa* increases growth and resistance to other disturbances. Forest Ecology and Management 480:118645.

White, C. A. 1985. Wildland fires in Banff National Park 1880-1980. National Parks Branch Occasional Paper 3, Parks Canada, Banff, Alberta, Canada.

Whitman, E., M.-A. Parisien, D. K. Thompson, and M. D. Flannigan. 2019. Short-interval wildfire and drought overwhelm boreal forest resilience. Scientific Reports 9:18796.

Williams, M. A., and W. L. Baker. 2011. Testing the accuracy of new methods for reconstructing historical structure of forest landscapes using GLO survey data. Ecological Monographs 81:63–88.

Yocom Kent, L. L., et al. 2017. Climate drives fire synchrony but local factors control fire regime change in northern Mexico. Ecosphere 8:e01709.

Yocom Kent, L. L., K. L. Shive, B. A. Strom, C. H. Sieg, M. E. Hunter, C. S. Stevens-Rumann, and P. Z. Fulé. 2015. Interactions of fuel treatments, wildfire severity, and carbon dynamics in dry conifer forests. Forest Ecology and Management 349:66–72.

Yocom, L. L., J. Jenness, P. Z. Fulé, and A. E. Thode. 2019. Previous fires and roads limit wildfire growth in Arizona and New Mexico, U.S.A. Forest Ecology and Management 449:117440.

Zald, H. S., and C. J. Dunn. 2018. Severe fire weather and intensive forest management increase fire severity in a multi-ownership landscape. Ecological Applications 28:1068–1080.

Supporting Information

Additional supporting information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/eap.2433/full